Habitat features of urban forest fragments supporting native lizards in the presence of introduced mammals

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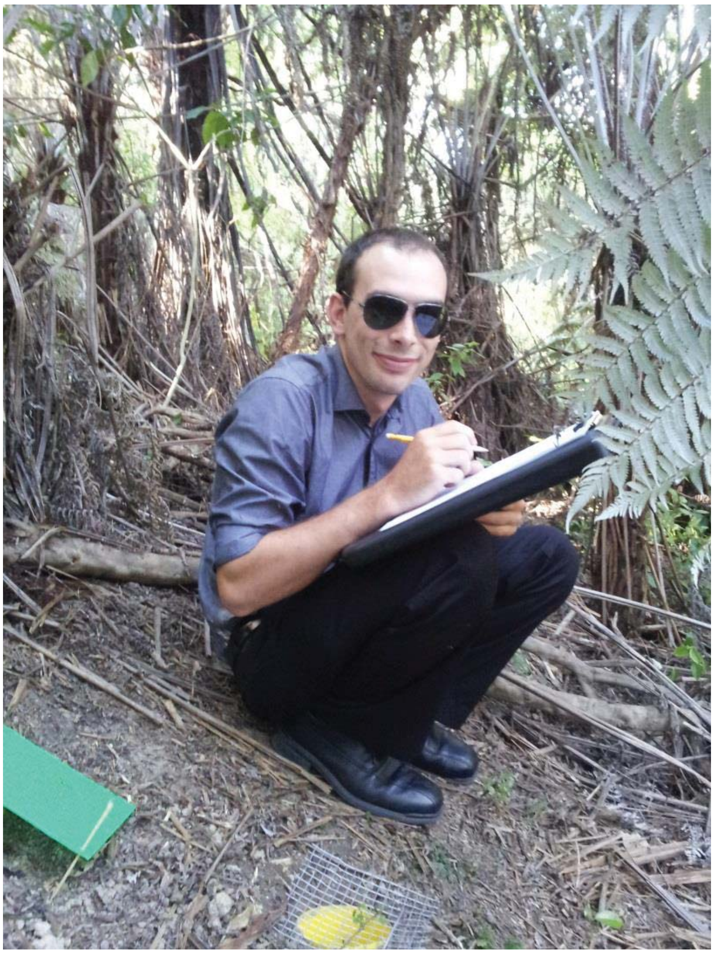

For Andy - the best field assistant anyone could ask for.

Abstract

Introduced species are responsible for declines and extinctions of native biota around the world, particularly on islands where native species are often more vulnerable to the effects of invaders due to a lack of shared evolutionary history. New Zealand's native lizards have suffered considerable range contractions, declines and extirpations as a result of predation and competition from introduced mammals, with some species being more vulnerable than others. Little is known about the mechanisms which allow some ground-dwelling native lizards to persist in the presence of introduced mammals. In this study, I describe the species composition and abundance of ground-dwelling lizard and introduced mammal assemblages in urban forest fragments, and investigate the relationship between them. I also describe the habitats used by native ground-dwelling lizards where introduced mammals are also present and investigate habitat features that may be important in promoting the coexistence of native lizards with introduced mammals. Finally, I compare various methods for surveying lizard (hand searching, artificial cover objects, pitfall traps) and mammal (tracking tunnels, snap traps) populations in urban forest fragments.

Estimating the proportion of tail loss can be used as a proxy to determine predation pressure on lizard populations. The rate of tail loss among urban lizards in this study was relatively high (41%), suggesting that these lizard populations are under considerable predation pressure. However, no relationship between the abundance of introduced mammals and native lizards was identified. Key features of the habitats supporting the highest abundance of native lizards in the presence introduced mammals include high canopy cover and high cover and structure of debris (leaf litter and branches/logs) in the lower shrub layer. Food availability in the form of invertebrate abundance does not appear to play a significant role in the coexistence of introduced mammals and native lizards, and the abundance of introduced mammals and exotic lizard competitors was not correlated with invertebrate abundance. Hand searching is the most efficient method for identifying lizards in urban bush fragments. ACOs and pitfall traps had only low efficiency in this study and are not recommended for future studies. I found that tracking tunnels may be an alternative to snap traps for indexing mammal abundance in urban environments where the risk of trapping non-target wildlife, pets and the public is high.

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Preface

Ko Pukeatua
Ko Te Wharau
Ko Tumupakihi
Ko Te Taoū
Ko Ngāti Whātua
Te Aute Te Awhe!

Hoinei te mana, hei maru mōku i te Kunenga ki Pūrehuroa kei Oteha

Nō reira, rātou ki a rātou
Tātou ki ā tātou
Kia ora mai tātou katoa

This is the pepeha[1] of the tāngata whenua whose rohe includes Massey University's Albany campus and several of the forest patches used in this study.

According to Maori kōrero, mokomoko are descended from Punga - the ancestor of sharks, rays, fish, insects, reptiles and all other things considered to be ugly or repulsive. Mokomoko were feared and often seen as omens of bad luck as they were thought to be representatives of Whiro, the god of darkness, evil and death. However, they were also seen as kaitiaki that would protect burial sites and important buildings. What follows is a re-telling of this kōrero describing the origin of lizards following the creation of the earth and sky.

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[1] A glossary of Maori terms can be found at the end of the preface

In the beginning there was no sky, no sea and no earth, only darkness. Papatūānuku, the earth mother, and Ranginui, the sky father, held each other in a tight embrace. They had many children, including Tangaroa (god of the sea), Tūmatauenga (god of war), Rongo-mā-Tāne (god of cultivated foods) and Tāne-mahuta (god of the forest). The children became frustrated with living in darkness between their parents and decided that their parents must be separated. Many of them tried and failed, but it was Tāne-mahuta who finally managed to push apart his parents, breaking their embrace and letting in the light.

Tāwhirimātea, the god of storms and winds, was angry at the separation of his parents, and vowed to his brothers that they would have to deal with his anger. He went to the heavens to join his father Ranginui and sent his children, the winds, to attack his siblings. Tangaroa fled to the sea to escape the onslaught. Tangaroa's son, Punga, had two offspring: Ikatere and Tū-te-wehiwehi (also called Tū-te-wanawana). Terrified by Tāwhirimātea's attack, Ikatere and Tū-te-wehiwehi had to decide where to go. Ikatere went to the sea with his children, the fish. Tū-te-wehiwehi chose the land and took refuge in the forest, becoming the ancestor of ngārara.

Glossary of Māori words:

Iwi – Tribe, extended kin. A large group descended from a common ancestor.

Kaitiaki – Guardian, custodian or protector.

Kōrero – Oral tradition.

Mokomoko – lizards, including geckos and skinks.

Ngārara – Reptiles, also includes the giant reptiles of Māori legends.

Pepeha – Introduction, tribal saying. A pepeha is a set of verses that describes a person’s links to a particular iwi and their links to the area and their ancestors.

Rohe – Territory, boundary. Often referring to the region a particular iwi identifies with.

Tāngata whenua – Indigenous people of the land.

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General Introduction

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1.1 – Evolution in the absence of mammalian predators

The interaction between predators and prey is a key component of natural selection in natural systems, with predators being under strong selection to detect and capture prey, and prey under strong selection to avoid being detected and captured. It is this reciprocal selection pressure that results in an evolutionary 'arms-race' between predators and their prey (Møller *et al.*, 2009; Genovart *et al.*, 2010). Many anti-predator strategies, while beneficial, may also carry a fitness cost – for example sockeye salmon (*Oncorhynchus nerka*) with smaller body sizes experience less predation by bears (*Ursus spp*) than larger salmon, but are also less successful than larger males when courting females, which is likely to reduce their overall fitness (Quinn *et al.*, 2001). Predator avoidance strategies that are associated with reduced fitness are costly to maintain and therefore may be modified or lost in the absence of predators (Blumstein & Daniel, 2002; Beauchamp, 2004; Blumstein & Daniel, 2005).

The degree of loss of predator avoidance traits due to lack of predation pressure from a specific predator is influenced by a multitude of factors including the phenotypic plasticity of the trait, the duration of isolation from the predator, and the presence of other predators in the environment (Blumstein *et al.*, 2004). As such, predator avoidance traits may show variable responses to isolation, with some being lost very quickly with the potential to re-emerge should the species be re-exposed to predators (Losos *et al.*, 2006; Jamieson & Ludwig, 2012), while other traits may take extended periods of isolation to disappear (Stankowich & Coss, 2007).

Islands can provide an extreme example of relaxed selection by predation, as they may be missing entire guilds of predators and as a result often have faunas with reduced predator-avoidance strategies and increased naivety which are especially vulnerable to the impacts of introduced predators (Case & Bolger, 1991; Blumstein & Daniel, 2005). The islands of New Zealand are a good example of this, having a long evolutionary history of isolation which began with the separation from Gondwana and opening of the Tasman Sea approximately 80 million years ago (Cooper & Millener, 1993; Neall & Trewick, 2008). New Zealand has been near its current position approximately 1500 km from the nearest major landmass of Australia and 1600 km South of New Caledonia for approximately 30 million years (Worthy & Holdaway, 2002; Neall & Trewick, 2008). To date, the fossil record has shown a remarkable lack of some major vertebrate groups including snakes and marsupials. Terrestrial mammals are also largely absent from the fossil record. Prior to the arrival of humans in New Zealand, the only terrestrial mammals were at least 7 species of bats (4 species known from fossils are yet to be described) and one small mouse-like mammal, which was recently discovered in sub fossil deposits in the South Island. (Worthy & Holdaway, 2002; Worthy *et al.*, 2006; Tennyson, 2010). As only 3 bone fragments have been found to date, its life history and phylogenetic affinities remain unclear but it is not closely related to any extant mammal groups. As such, it is unknown whether this mouse-like mammal was likely to be a predator of lizards but it is thought it became extinct as a result of severe climate change in the Middle Miocene period (16 - 11 million years ago).

In pre-human New Zealand, natural predators of ground-dwelling vertebrates were mainly large reptiles such as tuatara (*Sphenodon punctatus*) and avian predators such as morepork (*Ninox novaeseelandiae*) and kingfishers (*Todiramphus sanctus*) that mainly use visual cues to locate prey. The anti-predator strategies of native species reflect this, with many native reptiles and flightless birds displaying cryptic colouration and reduced flight behaviours (Daugherty *et al.*, 1993; Worthy & Holdaway, 2002; Hoare, 2006). As a result of its evolutionary isolation, the native fauna of New Zealand has evolved in an environment largely without predatory mammals for up to 16 million years depending on the time of extinction of the mouse-like mammal species. This isolation from mammalian predators has been postulated to explain the unusually high levels of flightlessness and gigantism in diverse groups of native species such as the large, flightless moa (Ratite) and weta (Orthoptera), and the fact that many species also have K-selected life strategies with low reproductive outputs and long lifespans – for example kakapo (*Strigops habroptilus*), *Powelliphanta* land snails and tuatara (Daugherty *et al.*, 1993; Cree, 1994). The morphological, behavioural and life-history traits that have resulted from New Zealand's long evolutionary isolation mean that native species are often ill-equipped and particularly vulnerable to predation by introduced mammals.

1.2 – Impacts of introduced species on native fauna

The introduction of alien species to new environments will often result in population declines and range contractions of native species through predation, competition, herbivory, habitat alteration, parasitism and/or the introduction of novel pathogens (Mack *et al.*, 2000; Davis, 2009). Among these, competition and predation are the most significant factors that contribute towards the loss of biodiversity as a result of the introduction of exotic species.

Introduced species can cause declines and extinctions of natives through competition, particularly for food resources. Introduced species may outcompete natives by foraging more efficiently (Kenward & Holm, 1993), or by excluding native species from utilizing food resources (Beggs & Wilson, 1991). For example, the Asian house gecko (*Hemidactylus frenatus*) is a highly competitive species that is responsible for the displacement and extinction of many native gecko species following its introduction to various islands in the Pacific and Indian Oceans. The Asian house gecko has been observed to outcompete native species by aggressively excluding natives from refuge and feeding sites (Case *et al.*, 1994; S. G. Brown *et al.*, 2002; Cole *et al.*, 2005), and by foraging more efficiently (Petren & Case, 1996).

Some of the most significant impacts of introduced species on native fauna occur by predation, often leading to population declines and extinctions (Iverson, 1978; Mack *et al.*, 2000; Towns *et al.*, 2006; Medina *et al.*, 2011). A recent study by Paolucci *et al.* (2013) was the first to show that the effects of introduced predators on native prey are much greater than the effects of native consumers. It is thought that a lack of adaptation to the impacts of introduced predators that were not present through native species' evolutionary histories is responsible for the more severe effects of introduced predators compared to native predators. Introduced mongooses (*Herpestes auropunctatus*), for example, are known to have serious impacts on native prey. Over the last 155 years, mongoose were found to be primarily responsible for up to 12 extinctions and 13 extirpations of herpetofauna from the islands of the West Indies (Henderson, 1992). But it's not just island species that are at risk, as introduced predators can also have serious consequences for native species on continents. A long-term fox (*Canis vulpes*) baiting experiment in Australia found that the density of native sand goannas (*Varanus gouldii*) increased five-fold following fox control, and other smaller scincid species were also observed to increase in density (Olsson *et al.*, 2005). While extinctions and population declines are important and conspicuous consequences of introduced predators, the loss of genetic diversity in populations that have undergone dramatic declines as a result of predation is another crucial impact that is largely poorly understood. Studies have suggested that lizard populations which have recovered following rat eradication may suffer depleted genetic diversity (Gasc *et al.*, 2010).

1.3 – Impacts of introduced species on islands

This lack of evolutionary history with some predator groups has long been implicated in the higher vulnerability of island biotas to introduced species. One study concluded that 95% of all extinctions in birds have occurred on islands, as well as 90% of extinctions in reptiles and 70% in mammals (Keitt *et al.*, 2011 but see Paolucci et al., 2013). The introduction of the brown tree snake (*Boiga irregularis*) to the island of Guam in the 1950s is a well-known example of the devastating effect introduced predators can have on islands. Within 40 years the introduced snake had caused the extinction or extirpation of at least 12 bird species and up to 5 reptile species, and was implicated in significant range contractions of many more species including bats (Savidge, 1987; Fritts & Rodda, 1998).

As a consequence of loss of species, biodiversity can be severely affected by alien invasions on islands due to the high levels of endemism and many relictual or archaic forms that are often found on islands (Daugherty *et al.*, 1993; Kier *et al.*, 2009). New Zealand, for example, is home to several ancient groups including Leiopelmatid frogs and the only living representative of the archaic order Sphenodontia, the tuatara (Daugherty *et al.*, 1993; Tennyson, 2010). The establishment of feral cats (*Felis catus*) on Stephen's Island in the 1890's is one example of the impact of introduced species on biodiversity. This island was home to the only remaining population of the endemic Stephen's Island wren *(Xenicus lyalli*), which was one of only 7 members of the ancient, endemic Acanthisittidae family, only two species of which remain. Both the Stephen's Island wren and the endemic Stephens Island Piopio (*Turnagra capensis minor)* became extinct following the arrival of cats on the island. The island biodiversity also declined significantly - at least 25 species of land bird were present on the island prior to the arrival of cats and now just 11 remain, many of which are common elsewhere (Galbreath & Brown, 2004; Medway, 2004).

Rodents are among the most successful non-human predatory invaders - they have reached 90% of islands in the world, negatively affected at least 160 taxa of plants and animals and have been implicated in at least 50 extinctions (Towns *et al.*, 2006; Donlan & Wilcox, 2008; Keitt *et al.*, 2011). Four species of rodent, Pacific rats/kiore (*Rattus exulans*), ship rats (*R. rattus*), Norway rats (*R. norvegicus*) and mice (*M. musculus*), have been introduced to New Zealand and have spread widely. Collectively, rats have invaded at least 142 offshore islands around New Zealand where they have been responsible for the extinction and decline of many groups of native species including invertebrates, reptiles, birds and plants (Towns *et al.*, 2006). One well-known example of the sometimes catastrophic impacts of introduced rats on island biodiversity is the 1962 invasion of Big South Cape Island by ship rats, where seven species in total became extinct. A bat and a wren species endemic to the island became globally extinct, while another bat, a flightless weevil, and five more bird species became locally extinct (Towns *et al.*, 2006).

1.4 – New Zealand's reptile fauna

New Zealand is home to three endemic reptile lineages: Sphenodontidae (tuatara), Diplodactylid geckos and Lygosomine skinks. Although New Zealand has low high-order diversity (having only few reptile families) it is one of the most species-rich temperate regions when corrected for area (Daugherty *et al.*, 1994). The reptile fauna of New Zealand is also highly unique: the tuatara is the only living representative of an ancient lineage; all but one of the gecko and skink species utilise the rare reproductive strategy of vivipary; and many gecko species are unusually long-lived and adapted to cold environments (Cree, 1994; Chapple *et al.*, 2009; Nielsen *et al.*, 2011).

Native skinks have evolved in relative isolation from mammalian predators since they arrived in New Zealand via over-water dispersal from New Caledonia approximately 16 – 22 million years ago, (Worthy & Holdaway, 2002; Chapple *et al.*, 2009; Tennyson, 2010). Native geckos have been isolated for even longer – approximately 40 million years based on the time of divergence from their Australian relatives (Nielsen *et al.*, 2011). It has been suggested that relaxed predation pressure over long periods such as these can lead to island species showing increased predator naivety and a lack of appropriate responses, and indeed this may be the case for New Zealand's herpetofauna (Case & Bolger, 1991; Blumstein & Daniel, 2005). The most common anti-predator behaviours of native lizards are cryptic colouration and behaviours and reduced escape behaviours, which are appropriate for the primarily visually oriented natural predators such as tuatara and kingfishers , but may be maladaptive in the presence of mammalian predators (Daugherty *et al.*, 1993; Worthy & Holdaway, 2002). Native species that use 'freeze' behaviours as an anti-predator strategy are likely to be an easy target for highly mobile mammalian predators such as rodents and mustelids which hunt using olfactory cues and can easily locate a motionless lizard.

Including eight non-resident native species (migrant or vagrant species such as sea snakes and turtles), one introduced species and two extinct native species, the total number of described reptile species in New Zealand is 69. Among the 58 resident native reptile species 51 are listed as at risk or threatened, with only 7 species classified as non-threatened (Hitchmough *et al.*, 2013). Of these 58 native reptile species, 43 are listed as either range-restricted or conservation-dependant (Hitchmough *et al.*, 2013), and approximately 40% of New Zealand's reptile species are either entirely or largely restricted to rat-free islands (Daugherty *et al.*, 1994; Towns & Daugherty, 1994). According to Hitchmough *et al.* (2013), a further 41 taxa have not yet been formally described and with further taxonomic studies it is likely that each will be confirmed to be a separate species, potentially bringing the total number of species in New Zealand to 110.

The mainland Auckland region is home to eight native lizard species: the copper (*Oligosoma aeneum*), ornate (*O. ornatum*), moko (*O. moco*), shore (*O. smithi*) and striped skinks (*O. striatum*) and the Auckland green (*Naultinus elegans*), forest (*Mokopirirakau granulatus*) and Pacific geckos (*Dactylocnemis pacificus*). The introduced rainbow skink (*Lampropholis delicata*) is also present in Auckland, and is common in suburban gardens (Gill & Whitaker, 1996). Additionally, translocated populations of Duvaucel's geckos (*Hoplodactylus duvaucelii)* and tuatara, and remnant populations of common geckos (*Woodworthia maculatus*) and egg-laying skinks (*O. suteri*) can be found on offshore islands in Auckland’s Hauraki Gulf, such as Tiritiri Matangi, Motutapu and Rangitoto (Gill & Whitaker, 1996; Baling *et al.*, 2013).

1.5 – Impacts of introduced mammalian predators on New Zealand native lizards

Since the arrival of humans in New Zealand, at least 31 mammal species have been introduced and subsequently established wild or feral populations (Parkes & Murphy, 2003). The first mammal introduction may have occurred as much as 2000 years ago, before permanent human settlement when Pacific rats (kiore) arrived with explorers visiting from the Pacific Islands, and certainly occurred when the first Maori colonised New Zealand approximately 800 years ago (the exact dates have long been disputed, see Hogg *et al.*, 2003; Wilmshurst & Higham, 2004; Wilmshurst *et al.*, 2008). Cats, weasels (*Mustela nivalis*), stoats (*M. erminea*), hedgehogs (*Erinaceus europaeus*) and rodents are among the 11 introduced mammal species which prey upon native fauna (Towns *et al.*, 2001). Introduced mammals have had an impact on a wide range of native taxa, and are responsible for the total extinction of 58 bird species, two frog species and one bat species, as well as the local extinctions of multiple plant and invertebrate species (Towns *et al.*, 2006; Towns *et al.*, 2013).

Native lizards have also suffered significant population declines, range contractions and extinctions as a result of predation by introduced mammals. Prior to the arrival of humans the reptile fauna of the North Island included tuatara, eight species of geckos and at least 14 species of skinks, five of which were widely distributed (Gill & Whitaker, 1996; Towns *et al.*, 2001; Towns *et al.*, 2006; Chapple *et al.*, 2008). Today the natural range of the tuatara is completely restricted to offshore islands (excluding translocated populations), only six gecko species remain, only two skink species are widespread on the mainland, and five skink species are largely restricted to offshore islands. Additionally, several lizard species are now locally extinct on the mainland, including the Duvaucel's gecko, Robust skink (*O. alani*), and McGregor's skink (*O. macgregori*). Predation by rats has been suggested as the main reason for the extirpation of these species (Towns & Daugherty, 1994; Gill & Whitaker, 1996; Towns *et al.*, 2001; Hoare *et al.*, 2007a; Chapple *et al.*, 2008).

As previously mentioned, rodents are among the most successful non-human mammal invaders, and are known to be responsible for declines and extinctions in many groups, including reptiles. Many examples of the impacts of introduced rodents have come as a result of studying how native reptile populations respond following rodent eradication (Towns, 2009). For example, prior to the eradication of Pacific rats from Korapuki Island, lizards were relatively less abundant when compared with nearby Middle Island which was rat-free. After rats were eradicated from Korapuki island, capture rates of shore skinks on rocky beaches increased up to 50-fold over the following 5 years, capture rates of common geckos increased and a translocated population of the rat-sensitive Whitakers skink (*O. whitakeri*) was observed to be breeding on the island (Towns, 1991; Towns & Daugherty, 1994; Towns, 1996). As well as increased density and reproductive success, reptile populations have been known to recolonise parts of their range after the eradication of invasive rats. The Fiordland skink, for example, was observed to recolonise Breaksea Island unaided, most likely from a nearby rock stack, within two years of Norway rats being eradicated from the island (Taylor & Thomas, 1993).

Some native lizard species appear to be more vulnerable to extinction than others, for a variety of reasons including degree of habitat specialisation, body size and geographic range size (Hitchmough *et al.*, 2010; Tingley *et al.*, 2013). Exposure to exotic predators and high human population densities were also found to be indicative of higher extinction risk, and these factors are particularly relevant for urban lizard populations. Ground-dwelling speciesmay also experience increased extinction risk asspecies such as skinks can be more vulnerable to predation by exotic mammals due to their limited ability to climb (Towns & Daugherty, 1994; Hitchmough *et al.*, 2010 but see Tingley et al., 2013).

Although predatory mammals have had a severe impact on the abundance and distribution of many lizard species, there are some species that are able to persist in the presence of mammalian predators (Towns, 1991). One such example is the lizard assemblage at Pukerua Bay, where common geckos, common skinks (*O. polychroma*), brown skinks (*O. zealandicum*) and copper skinks coexist (although some species only exist at low densities) with a diverse suite of mammalian predators including ship rats, mice, cats , weasels, stoats, and hedgehogs (Towns & Elliott, 1996; Hoare *et al.*, 2007a). While some lizard species may be able to persist with introduced mammals for extended periods, for some species it is not sustainable in the long term. For example, the last known mainland population of Whitakers skinks has been studied at Pukerua Bay from 1982 to 2006. During the last six years of monitoring (2000 – 2006) just two individuals were captured and it was determined that the population faced imminent extinction, mainly as a result of predation and despite efforts to reduce predation pressure (Towns & Elliott, 1996; Hoare *et al.*, 2007a).

Those lizard populations that do manage to persist in the presence of introduced mammals often undergo a variety of behavioural and demographic changes, as was found in a study on Ohinau Island. It was found that prior to the eradication of the only introduced predator on the island – the Pacific rat - the native Duvaucel's gecko population suffered reduced recruitment, was skewed towards larger individuals, and showed altered habitat selection in order to avoid predation (Hoare *et al.*, 2007b). This study is one of very few investigating the habitat features that allow native lizards to coexist with introduced predators. It is largely unknown which factors may contribute towards the coexistence of these species.

1.6 - The need for alternative methods of managing native lizard populations.

Current practices for the management of native lizard populations in New Zealand typically involve lethal control of mammalian predator populations, either with the goal of eradication or reduction of predator population densities. Eradication has been proven to be very effective on islands, such as Korapuki Island (Parkes & Murphy, 2003). The capture rates of shore skinks and common geckos on this island increased up to 30-fold following the eradication of rats (Towns *et al.*, 2006). On Ohinau Island, rat eradication led to increased capture rates and habitat use by Duvaucel's geckos (Hoare *et al.*, 2007b). While the limited size of many islands may allow complete eradications to be achieved, the much larger area and typically more complex land use patterns of the mainland mean that control rather than eradication of introduced mammal populations is the goal at most mainland locations (Thomas & Taylor, 2002; Towns *et al.*, 2006; Howald *et al.*, 2007; Keitt *et al.*, 2011).

The use of lethal traps and poisons in bait stations are two common methods of controlling mammal populations in New Zealand. While effective, these methods must be implemented properly in order to avoid the development of trap- or bait-shyness in target species or the killing of non-target species. Other negative aspects of these methods are that they are highly labour-intensive due to the time and effort required in setting and checking traps, or in maintaining bait stations, often over large areas (Mack *et al.*, 2000; Parkes & Murphy, 2003). Less labour-intensive methods such as aerial distribution of poison bait are becoming more common, but this approach lacks target specificity and therefore is more commonly used for eradication attempts on islands. Ethical issues, public health concerns and public opposition mean that aerial distribution of poison bait is not feasible for use in some parts of the mainland such as urban areas and farmland where human health and non-target animals such as pets and cattle may be at risk (Fitzgerald *et al.*, 1996; Green & Rohan, 2012).

A second factor to consider when controlling introduced mammals on the mainland is the typical patchiness of control efforts. Time, resource and budget constraints dictate that specific ecologically significant areas are targeted for mammal control efforts, which then require on-going management due to the constant threat of re-invasion from surrounding, un-managed areas (Zavaleta *et al.*, 2001). This effect may be exacerbated in urban areas where private property may provide a refuge from poisoning and trapping programmes to control mammalian predators. The many issues with using lethal traps and poisons mean that it is not feasible as the sole method of reducing the impacts of mammalian predators in urban areas and additional measures need to be developed.

A second method of managing native lizard populations is by habitat modification to provide protection from predators, food resources, or a combination of both. While the large-scale impacts of introduced mammals on native lizard populations have been well documented, there has been little research into the mechanisms that may allow native species to coexist with introduced mammals (Towns *et al.*, 2006). The use of habitat manipulation as a method of protecting native lizard populations has been much less successful than lethal control methods (Lettink *et al.*, 2010). In the 1980’s an attempt was made to restore the habitat of the last remaining mainland population of the Whitaker’s skink (*O. whitakeri*) with the intention of providing protection from introduced mammalian predators. Grazing stock was removed from Pukerua Bay Scientific Reserve and re-vegetation was attempted but was unsuccessful, leading to a rapid increase in rank introduced grasses and other weeds (Hoare *et al.*, 2007a). It is thought that the seeds from these introduced plants supported population explosions of rodents, which are known to prey on lizards. Increased rodent populations can also support larger populations of other mammalian predators such as mustelids and cats, which can switch prey to lizards when rodent populations decrease. Capture rates of both the Whitaker's skink and the copper skink declined between 1984 and 2006, to the point where just two Whitaker's skinks were caught in the last 6 years of sampling and the population faced imminent extinction. This example illustrates the difficulties of predicting lizard responses to habitat manipulation and that a clearer understanding of lizard habitat requirements and the complex interactions with predators is required for effective management.

Additionally, many urban lizard habitats are threatened and in decline due to ongoing development, particularly in the Auckland area. Development in urban areas is often mitigated by re-vegetation, which may provide unsuitable habitats for lizards. Often there are large gaps between plants, plants are very small, there is little to no ground cover, and wood chips or mulch are typically used, which suppresses any undergrowth. As a result, re-vegetated areas often do not provide enough protection from predators and are highly unlikely to support populations of native lizards.

New Zealand's long evolutionary isolation has resulted in a fauna that is susceptible to the impacts of introduced mammalian predators. Given the effects of introduced species on natives, especially island-dwelling natives, it is not surprising that introduced mammals have had significant impacts on native lizard populations in New Zealand. The current methods of managing native lizard populations in the presence of introduced mammals are not always appropriate, and therefore other approaches must be developed. My study aims to determine which habitat variables may enable native lizards to co-exist with introduced mammals; hence my findings could have implications for future management of lizard habitats andre-vegetation efforts.

The overall aim of this study was to identify habitat features which promote the coexistence of native lizards with introduced mammals in urban forest fragments.

My primary objectives were to:

1) Describe the species composition and abundance of ground-dwelling lizard and introduced mammal assemblages in urban forest fragments, and investigate the relationship between them.

2) Identify and describe the types of habitats used by native ground-dwelling lizards where introduced mammals are also present. The floristic composition, vegetation structure, invertebrate abundance and composition of these habitats will be documented.

3) Determine which habitat features may be important in promoting the coexistence of native lizards with introduced mammals.

4) Compare various methods for surveying lizard (hand searching, artificial cover objects, pitfall traps) and mammal (tracking tunnels, snap traps) populations, and compare different baits used for invertebrate pitfall trapping.

1.7 – Thesis structure

The aims of my thesis are addressed in the following chapters:

Chapter 1 provides a general introduction to the evolutionary history of isolated fauna and the effects of introduced predators on native species.

Chapter 2 describes the native lizard and introduced mammal populations of urban forest fragments in Auckland, New Zealand. The relationship between native lizard and introduced mammal populations is investigated, and the efficiency of common survey methods compared.

Chapter 3 describes the structure and composition of habitats used by native lizards, and the effects of habitat on native lizard abundance.

Chapter 4 describes the invertebrate community of habitats used by native lizards, and investigates the effects on lizard abundance. The potential effects of competitors and repeated sampling on invertebrate communities is also investigated, and the use of baits in invertebrate pitfall sampling analysed.

Chapter 5 summarizes the main findings of this study and discusses future research directions stemming from this project.

Chapter 2

Abundance and distribution of lizards and mammalian predators in urban forest fragments

Plate 2.1 - Ship rat (*Rattus rattus*) seen climbing a tree fern in the Eskdale study site. Rats are known to prey on native lizards as well as other indigenous fauna such invertebrates, birds, chicks and eggs.

2.1 – Introduction

2.1.1 – How native lizards survive in the presence of introduced mammals

Despite the significant impacts of introduced predators on native reptiles, some species are able to coexist with introduced predators. Rats, mice, cats and hedgehogs are widespread on the mainland, particularly in urban areas, where some native lizards such as the copper skink and ornate skink are also present. However, the mechanisms by which some native lizard species, particularly ground-dwelling lizards, can coexist with introduced predators are largely unknown. While exotic lizard species may be able to cope with the predation pressure of introduced mammals due to their higher reproductive rate, this tactic is highly unlikely to be used by native reptiles which are typically slow to reach sexual maturity and produce small clutches once per season or less (Cree, 1994). Common geckos (*Woodworthia maculata*) in the Central Otago region, for example, have a very low reproductive output of just 0.85 offspring/female/year as they often have clutch sizes less than two and only reproduce biennially (Cree, 1994). In general, skinks are slightly more productive than geckos, with an average reproductive output of one to five offspring/female/year, although some species appear to reproduce only biennially (Cree, 1994). This is not the case for the exotic rainbow skink, which despite living in the same predator environment as native species, is able to reach much higher population densities, possibly due to its ability to produce up to eight eggs per clutch and three clutches per year, although this does vary between populations and is typically two to five eggs per clutch (Peace, 2004). Another explanation for the greater success of rainbow skinks in the presence of predators is their evolutionary history with a variety of predators in their native range, including mammals and other large, mobile predators such as snakes, goannas (*Varanus spp*) and quolls (*Dasyurus spp*).

It is most likely that native lizards coexist with introduced mammals through a combination of behavioural avoidance tactics and the use of refugia to avoid predation events. In a pre- and post-eradication study on Ohinau Island, it was found that although Duvaucel's geckos and Pacific rats use the same habitats on a larger scale, there is little overlap at the micro-habitat scale (Hoare *et al.*, 2007b). After the rat eradication, and before any recruitment could have taken place, Duvaucel's geckos were observed to use a greater proportion of the habitats previously preferred by rats, indicating that these geckos are able to coexist with introduced rodents by behaviourally avoiding habitats preferred by rats. In addition, two more lizard species (common gecko and copper skink) that had not been found on the island in more than 30 years were re-discovered soon after the eradication of the rats and prior to any recruitment, further suggesting that these species are capable of behavioural and habitat use plasticity in order to survive in the presence of rats (Hoare *et al.*, 2007b).

While the study by Hoare *et al.* (2007b) provides valuable insight on how geckos can alter their habitat use in order to survive with rats, there have been no studies on how skinks are able to survive when rats are present. Skinks are largely ground-dwelling and have a limited ability to climb and make use of vertical habitat, therefore it is likely that they could be more strongly affected by the presence of rats, and in fact primarily ground-active species have been found to be associated with higher extinction risk in native reptiles

(Towns & Daugherty, 1994; Hitchmough *et al.*, 2010). The fact that copper skinks and common geckos were not found on Ohinau Island for more than three decades prior to rat eradication (Hoare *et al.*, 2007b) suggests that these species may be severely impacted by rat presence, to the point that they cannot be detected. Some potential explanations for this include that these lizard species are so heavily preyed upon that their population densities are suppressed to below detectable levels, or that these lizard species alter their habitat use so dramatically that they are not detected.

2.1.2 – Coexistence of lizards and introduced mammals in urban forest fragments

The survival of native species with introduced predators is further complicated by the pressures of urban environments. Urbanisation typically results in fragmentation of natural habitats, changes in food resources, increased human disturbance, pollution, and alteration of predator communities (McKinney, 2002; Shochat *et al.*, 2004; Chace & Walsh, 2006; Rodewald *et al.*, 2011). Not surprisingly, these changes are greater than many species are able to tolerate, often resulting in a loss of biodiversity (McKinney, 2006; Shochat *et al.*, 2010; Sih *et al.*, 2011). Urban living is known to have an impact on lizard populations, affecting their movements, distribution, and morphology (Germaine & Wakeling, 2001; Brehme *et al.*, 2013; Lazic *et al.*, 2013). In New Zealand, the pressures of living in highly fragmented urban landscapes are known to have negative impacts on population size and distribution of native lizard species (Towns *et al.*, 2001; van Heezik & Ludwig, 2012). Based on subfossil deposits, the native reptile community of the Auckland region probably included tuatara, Duvaucel's geckos, common geckos, robust skinks, and Whitaker's skinks

(*O. whitakeri*) (Towns & Daugherty, 1994). Today, these species are no longer present in urban areas and throughout much of the mainland, possibly due to habitat loss, increased human disturbance and altered predator regimes that came with human settlement and urbanisation of Auckland. Only three native species are now widespread in urban areas: the copper and ornate skinks and the forest gecko. A further five species are found only at isolated locations within the greater Auckland region, generally away from urban areas (Gill & Whitaker, 1996; Towns *et al.*, 2001; Towns *et al.*, 2006; Chapple *et al.*, 2008). The introduced rainbow skink is also present in Auckland, and is common in suburban gardens (Gill & Whitaker, 1996).

One of the major threats to urban lizards in New Zealand is introduced mammalian predators. As with most urban areas, the Auckland area has a diverse suite of introduced mammalian predators including ship rats, Norway rats, mice, cats and hedgehogs. This study focusses on the rodent species, as they pose a considerable risk to lizard fauna (section 1.5) and are typically difficult to eradicate from urban areas (section 1.6). Both rat species are nocturnal, and occupy a wide range of habitats including forests of all sizes, wetlands, grasslands, industrial areas and rubbish dumps (Innes, 2005a, 2005b). Rats are known to prey on native lizards, and have been implicated in many declines and extinctions of native lizard populations (Towns & Daugherty, 1994; Towns *et al.*, 2001; Towns, 2009). Although not as much of a threat to native reptiles as rats, mice are known to prey on reptiles and have been shown to have a negative impact on some populations (Newman, 1988, 1994; Wedding, 2007; Norbury *et al.*, 2013), so they were also considered in this study. Hedgehogs have been found to prey upon lizards; one study found that skinks comprised 14% of hedgehog diets, while another found that skink captures declined with increasing hedgehog density (Jones & Norbury, 2010; Jones *et al.*, 2013). I also considered hedgehogs because in some habitats, hedgehog populations can reach relatively large densities, and given their widespread distribution even a relatively low level of predation on skinks could have significant effects (Reardon *et al.*, 2012). Although cats are important predators in urban ecosystems, they were not considered in this study for a number of practical reasons (see section 2.4.3).

2.1.3 – Surveying in urban forest fragments

Common methods for surveying lizards in New Zealand include live trapping (pitfall or funnel-type), systematic searching (may be done during the day or at night by spotlighting) and using artificial cover objects (hereafter ACOs). Pitfall trapping is routinely used in lizard surveys in New Zealand (Whitaker, 1967; Patterson, 1992; Lettink & Seddon, 2007; Norbury *et al.*, 2009; Hare, 2012b), however, it does have some key limitations including the considerable time required to set up a trapping grid and check traps daily, problems with unequal capture probabilities, and increased predation risk for lizards caught in traps (Newman, 1994; Towns & Elliott, 1996; Hare, 2012b). The use of ACOs is a comparatively new method but it is quickly becoming one of the most common techniques used for lizard surveys as they are relatively inexpensive and easy to set up, don't require daily checks, and allow lizards to freely come and go, which reduces predation risk (Hare, 2012a, 2012b; Lettink, 2012). Many published studies using ACOs on mainland New Zealand have been undertaken in open habitats such grassland, pasture, or duneland (D. J. Wilson *et al.*, 2007; Hoare *et al.*, 2009; Lettink *et al.*, 2010; Lettink *et al.*, 2011; Norbury *et al.*, 2013). Very few studies have focussed on forested areas, although native lizards are known to inhabit these environments (Gill & Whitaker, 1996). It has been suggested that ACOs may be less effective in shaded environments (such as forests) because their thermal properties attract lizards, although their structural attributes have also been found to be important (Thierry *et al.*, 2009; Lettink, 2012).

Studies regarding surveying techniques for small mammals such as rats in urban forest fragments are also scarce. Extensive research has been done on small mammals in New Zealand, but mainly in large tracts of forest away from urban areas (King *et al.*, 1996a; King *et al.*, 1996b; Efford *et al.*, 2006; Watkins *et al.*, 2009, 2010; Goldwater *et al.*, 2012). Lethal snap traps are a common and widely used method for obtaining indices of rodent abundance, but there are some important limitations including the time required to check and re-bait traps daily, and the capture of non-target species (King *et al.*, 1996b; Efford *et al.*, 2006; Theuerkauf *et al.*, 2011; Gillies, 2013). In urban areas, the problem of trapping non-target species is of greater concern as pet cats, dogs, or even children playing in the forest may be injured by snap traps. While the use of wooden or mesh covers over snap traps is recommended practice to reduce incidence of non-target captures (Gillies, 2013), it does not completely prevent them and the risk of unwanted bycatch may be considered to be too great to warrant using snap traps in some circumstances. The use of tracking tunnels has been suggested as a potential alternative to using snap-traps for obtaining indices of rat abundance by two different studies in New Zealand forests (K. P. Brown *et al.*, 1996; Blackwell *et al.*, 2002).

Considering the number of serious declines and extirpations of some native lizards caused by introduced mammals, it raises the question of exactly how other lizard species are able to persist. While the behaviour of semi-arboreal geckos in the presence of rats has been investigated (Hoare *et al.*, 2007b), the mechanisms behind the persistence of ground-dwelling lizards such as skinks in the presence of introduced mammalian predators remain unclear. In this part of my study, I will describe the abundance, diversity and morphometrics of ground-dwelling lizard assemblages, and the abundance and diversity of introduced mammals at six urban forest fragments. The information on lizard and mammal abundances will then be used in subsequent chapters to assess the relative importance of various habitat features for facilitating the coexistence of lizards with introduced mammals. The lizard and mammal communities at the six sites will be compared in order to assess the effects of predator populations on native lizards. As there is little information on appropriate survey methods for use in urban forest fragments, I will also compare the relative efficiency of various lizard surveying methods in urban forest fragments, and investigate the suitability of tracking cards as an alternative to snap-trapping for surveying small mammal populations in urban settings.

2.2 – Methods

2.2.1 – Study species

Ground-dwelling lizards

This study focusses on the native, ground-dwelling lizard fauna of urban Auckland, which is made up of two species – the copper skink and the ornate skink. Although it is not a target species, the introduced rainbow skink is also present in urban Auckalnd. The copper skink is New Zealand's smallest native skink (up to 62 mm snout-vent-length, SVL), and typically inhabits forest and scrub as well as suburban gardens that provide sufficient cover such as rocks, logs, bark and low-growing vegetation (Gill & Whitaker, 1996). The copper skink is a widespread species and is classified as 'not threatened' under the New Zealand Threat Classification System (NZTCS) (Hitchmough *et al.*, 2013). The ornate skink is larger (up to 80 mm SVL) and has similar habitat preferences to the copper skink, but is less common in suburban gardens (Gill & Whitaker, 1996). Under the NZTCS the ornate skink is listed as 'at risk' under the category 'declining' (Hitchmough *et al.*, 2013). The two species can be distinguished by the difference in body size, with the ornate skink typically being larger and more heavily built. The ornate skink also has a characteristic white 'tear-drop' marking beneath the eye (plate 2.2) and is usually mottled with large, pale blotches on its back and tail. The copper skink usually lacks these large pale markings and instead has a bright copper sheen with dark dorsolateral lines at the edges of its back (plate 2.2).

Plate 2.2 - Skink fauna of urban Auckland. Left: copper skink, *Oligosoma aeneum* (photo by Luca Butikofer). Middle: ornate skink, *O. ornatum*. Right: rainbow skink, *Lampropholis delicata*.

The rainbow skink is the smallest of the three skink species (up to 55 mm SVL), and is New Zealand's only introduced lizard. If caught, it is easily distinguished from native species by the large central scale on its head, where native species have two scales. Other distinguishing characteristics include its small size, comparatively long tail, slight build, and its rainbow coloured iridescent sheen. It is thought that it was introduced to New Zealand in the 1960's, most likely in cargo from Australia and since then it has spread to other parts of the North Island, probably in freight. The rainbow skink is a habitat and diet generalist species, and inhabits a wide range of habitats from forest and grassland more disturbed areas such as urban gardens, rank vegetation and industrial sites. Unlike most native species, this introduced skink is oviparous (egg-laying) and each female will deposit two to five eggs per clutch, sometimes communally, and up to three clutches per year (Gill & Whitaker, 1996; Peace, 2004).

Mammalian predators

The more common ship rat (120 - 160 g) is smaller than the Norway rat (200 – 300 g) and is an agile climber, often feeding and nesting in trees (Innes, 2005a, 2005b). The Norway rat climbs much less readily but is a strong swimmer and is usually associated with streams, swamps and other water features. Reproduction of rats varies greatly by site, but as an indication both species are capable of breeding year-round and typically have 6-8 offspring per litter (Innes, 2005a, 2005b).

Mice (up to 17 g) are also present throughout the Auckland area in forests, pasture, rubbish dumps and buildings, although they tend to be less abundant in areas where rats are also present due to predation by, and competition with the larger rodents (King *et al.*, 1996b; Ruscoe & Murphy, 2005; Goldwater *et al.*, 2012; Bridgman *et al.*, 2013). Mice are also capable of producing large numbers of offspring per year and can reproduce year-round in New Zealand. A study of a mouse population in a North Island forest found pregnant females in all seasons, with litter sizes of five to eight in spring/summer and one to five in autumn/winter (King *et al.*, 1996a).

Hedgehogs (up to 800g) are now widespread in New Zealand, including Auckland, and inhabit a wide range of habitats from native forest to pasture and urban gardens (Parkes, 1975; Moss & Sanders, 2001). Insects make up a significant portion of the diets of hedgehogs, but chicks, eggs, skinks and small geckos are also known to be common in hedgehog diets (Moss & Sanders, 2001; Jones *et al.*, 2005; Jones & Norbury, 2010).

2.2.2 – Study sites and sampling techniques

The Auckland region is located in the upper North Island of New Zealand, covers approximately 5,600 square kilometres and includes New Zealand's largest city, Auckland, with a population of 1.4 million in 2009 (McClure, 2012b). The region is located on a narrow isthmus between the Waitemata and Manukau Harbours and has a temperate, maritime climate characterised by warm, humid summers, mild winters and high rainfall year-round (McClure, 2012d, 2012a). Prior to human settlement, the majority of the area was covered with native Kauri (*Agatha australis*) and mixed conifer-broadleaf forests (McClure, 2012a). The clearing of Auckland's forests began with the settlement of the indigenous Maori approximately 700 years ago, but the majority occurred following European settlement of the area in 1840 (McClure, 2012a, 2012e). By the early 1900's, most of the forests had been felled and land use in the area began to change to pasture, which remains a dominant land use in the Auckland region (McClure, 2012c).

The six sites used in this study are within urban bush fragments in public parks and reserves across the northern part of Auckland city (figure 2.1). Bush fragments were chosen for several reasons: lizards are known to inhabit these habitats; long-term access to public reserves is more feasible than using private land; and public reserves are the most common areas managed by government and community groups for restoration of wildlife habitats.

Figure 2.1 - Map showing the locations of the six study sites within the northern part of the greater Auckland Region. Photo: Auckland Council GIS viewer.

Sites were selected within an 11 km radius in order to reduce geographic variation in climate and vegetation. Selection criteria were as follows: sites were large enough to accommodate two 50 x 50 m sampling grids and the vegetation was native mixed podocarp-broadleaf forest (whether regenerating or intact). Common tree species include *Cyathea dealbata* (silver fern), *Kunzea ericoides* (kanuka), *Phyllocladus trichomanoides* (Tanekaha or celery pine), *Myrsine australis* (red matipo), *Dicksonia squarrosa* (wheki or rough tree fern) and *Agathis australis* (kauri). The sites were bordered by various urban features including housing, schools, car parks, shops and businesses.

Eskdale Reserve

Approximately 45ha in size, this is the largest of the six sites in this study and is among the largest forest remnants on Auckland's North Shore. Eskdale Reserve is almost entirely surrounded by suburban houses, except on the southern edge where it borders a cemetery (visible as a grassed area between the reserve and the road at the bottom of figure 2.2).

Figure 2.2 – Aerial photograph of Eskdale Reserve. Red line indicates the reserve boundary. Blue line represents approximately 500m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

Kauri Glen

Approximately 26ha in size, but this value does not include several bush areas to the north and east which are not included within the legal boundary of the reserve but likely add to the total habitat available. Kauri Glen is mostly surrounded by suburban houses and on the southern side the reserve backs on to the sports field of a school (bottom centre of figure 2.3), while to the southeast there is a small light industrial area comprised mainly of offices.

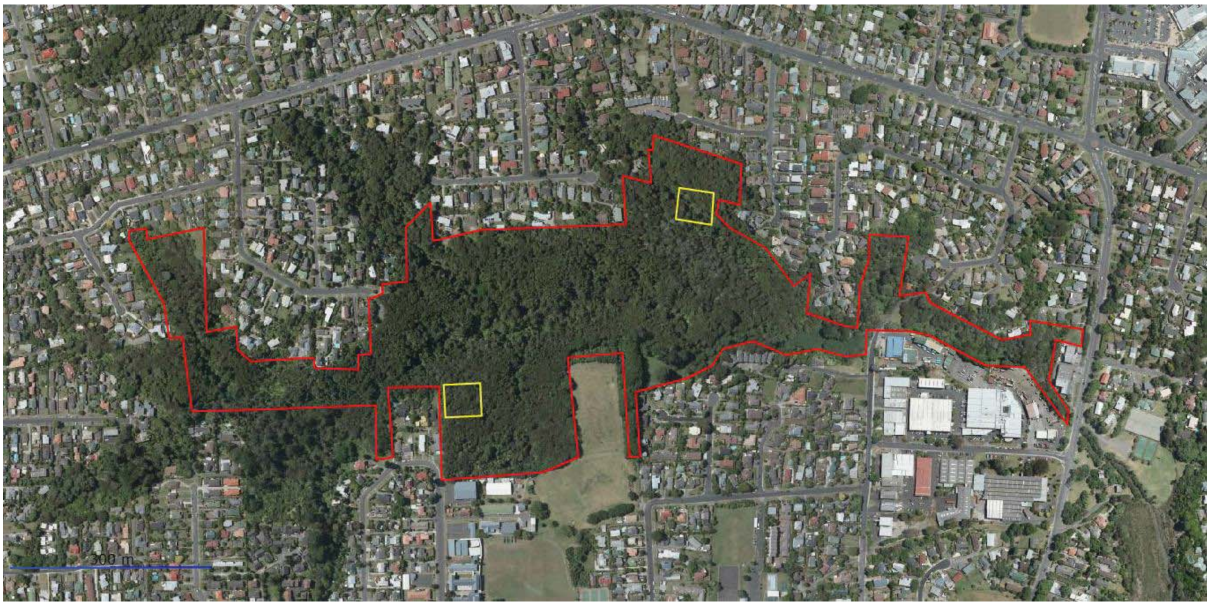

Figure 2.3 – Aerial photograph of Kauri Glen. Red line indicates the reserve boundary. Blue line represents approximately 300m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

Kauri Park

Approximately 22ha in size, and like most of the other reserves Kauri Park is mainly surrounded by suburban housing. There is a conference venue on the northeast edge of the reserve and an estuary which joins the Waitemata Harbour on the southwest edge of the park (figure 2.4). There are many other bush remnants nearby, some of which are similar in size to Kauri Park.

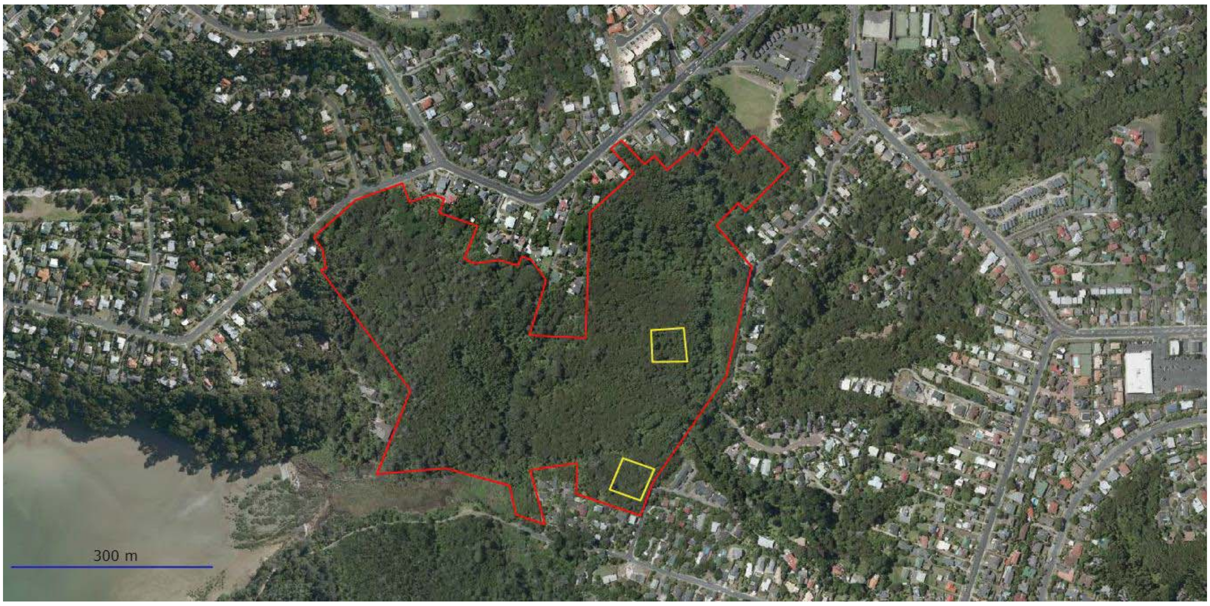

Figure 2.4 - Aerial photograph of Kauri Park. Red line indicates the reserve boundary. Blue line represents approximately 300m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

Lady Phoenix Reserve

This reserve is approximately 12ha in size. The reserve is mostly surrounded by suburban homes, although due to the location near an estuary to the west and semi-rural land to the east there are fewer homes nearby compared with most other sites. A stream runs along the southern edge of the reserve towards the estuary and the Waitemata Harbour. There are a few bush remnants and areas of scrub within the semi-rural land to the east but the matrix between these remnants is mainly pasture.

Figure 2.5 - Aerial photograph of Lady Phoenix Reserve. Red line indicates the reserve boundary. Blue line represents approximately 300m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

Silverdale Scenic Reserve

Approximately 5ha in size this is the smallest site used in this study, however it has the most diverse land use in the surrounding area. There are only a few homes directly adjacent to the reserve, all on the northern side. Although the western side of the reserve backs on to rural land including some areas of bush and scrub, this site is probably the most isolated in terms of proximity to other bush remnants. To the northeast is a small industrial area, while to the south is a retail area. It is worth noting that this area has undergone significant development in recent years. The aerial photograph is outdated and does not show the large shopping centre to the east and northeast of the reserve, respectively. The small triangle of land directly adjacent to the eastern boundary of the reserve has also been developed and is now a retail area. Additionally, the vacant land seen in the photograph to the north of the reserve is being converted into a suburban housing development.

Figure 2.6 - Aerial photograph of Silverdale Scenic Reserve. Red line indicates the reserve boundary. Blue line represents approximately 300m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

Torbay Heights

Approximately 10ha in size, this reserve is mostly surrounded by suburban homes. To the west and northwest are large areas of bush and scrub, although they are separated from Torbay Heights by major roads. There are also small bush remnants within the suburban area.

Figure 2.7 - Aerial photograph of Torbay Heights. Red line indicates the reserve boundary. Blue line represents approximately 300m for scale purposes. The approximate locations of the surveying grids are marked with yellow squares. Photo: Auckland Council GIS viewer.

At each site two surveying grids were set up. Starting points of the survey grids were chosen randomly, with the only constraints being that one side of at least one survey grid per site was within 10 m of the forest edge (including edges created by walking tracks) in order to included edge habitat as well as interior, and that all sampling points had to be at least 30 m from any stream (as per the Department of Conservation research permit). Survey grids were set up at least 2 m from the forest edge to avoid disturbance of survey equipment by members of the public. Each 50 m x 50 m grid consisted of three transects 25 m apart, and each transect had three sampling points 25 m apart (nine points per grid). At each sampling point, one tracking tunnel and three artificial cover objects (ACOs) were installed within a 1.5 m radius of the sampling point (plate 2.3).

Plate 2.3 - Part of a sampling point, showing a tracking tunnel (bottom left) and one of the three ACOs (top right).

After the grids were set up, they were left undisturbed for at least three to four weeks to allow animals to become used to the equipment (Lettink, 2007; Hoare *et al.*, 2009). As a result of the low capture rates achieved using ACO's, three pitfall traps were also installed at each sampling point within 0.5m of each other prior to the beginning of the main survey period (January to April 2013) and again the sites were left undisturbed for at least three weeks before surveys were conducted.

2.2.3 – Pilot surveys

In order to identify sites that were suitable for my study, ten potential sites were pre-surveyed from December 2011 to April 2012, and September to December 2012 to confirm the presence of rats and native skinks as the minimum requirements for mammals and lizards, respectively.

First, ACOs were checked every three to five days for a month at each site during initial surveys. Low capture and detection rates were achieved using ACOs and as a result only two of the ten potential sites (Torbay Heights and Silverdale Scenic reserve) were confirmed to have native lizards present by October 2012. To establish whether native lizards were present at the remaining sites, hand searching was carried out in October and November 2012. . Lizard data from ACO checks in the initial surveys were included in analyses to increase the number of lizard captures per site.

Initial rat surveys were carried out after lizard presence was confirmed at each site in order to avoid interfering with the lizard surveys. Baited tracking cards were changed daily for three nights at each grid. Note that rat tracking data from initial surveys are not included in analyses due to problems with rats locating more tunnels each night when baited cards were left out over consecutive nights (Appendix I).

Six sites with both native lizards and introduced rats present were selected as study sites: Eskdale Reserve, Lady Phoenix Reserve, Kauri Park, Kauri Glen, Silverdale Scenic Reserve and Torbay Heights (figure 2.1).

2.2.4 –Lizard surveys

The main survey period occurred between January and April 2013 in order to coincide with warm temperatures in the Austral summer that are associated with higher lizard activity. The main survey period comprised three sampling sessions, with each session lasting approximately 10 days (figure 2.8). Two of the six sites could be surveyed simultaneously, meaning each site was left undisturbed for a minimum of 14 days between sessions while the other sites were surveyed.

Pitfall traps were not originally intended to be used for lizard surveys, due to the limitations mentioned in section 2.1.3. However, due to the low capture rates achieved with ACOs in the initial surveys, the design of pitfall traps originally intended for invertebrates only was modified to allow lizards to be captured to augment captures using ACOs which were also used in the main survey period. Pitfall trapping was carried out on three occasions at each site, with each trapping session lasting two nights, giving a total of 1,944 trap nights.

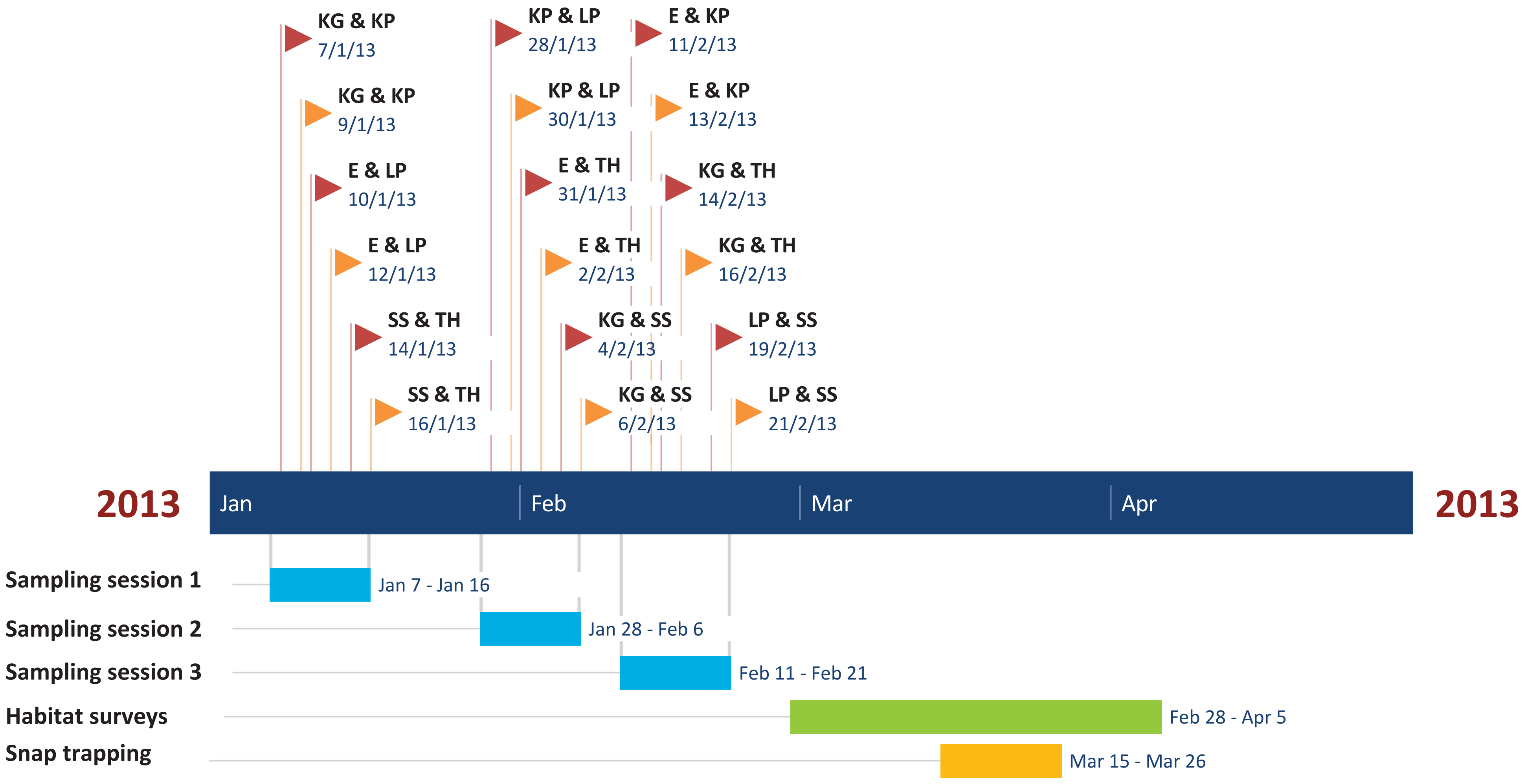

Figure 2.8 - Timeline for the main survey period, January – April 2013. ▶ indicates the start date of pitfall trapping & ACO checking at each site for each sampling session. ▶ indicates the date of mammal tracking at each site for each sampling session. E = Eskdale Reserve, KG = Kauri Glen, KP = Kauri Park, LP = Lady Phoenix Reserve, SS = Silverdale Scenic Reserve, TH = Torbay Heights.

However, after three pitfall trapping sessions were completed at each site, it became clear that the traps were highly inefficient (table 2.4) and so few lizards were caught that it was not possible to assess the influence of bait type on lizard captures. A lack of experienced researchers and strict time limitations meant it was not possible to carry out more extensive hand searching to increase lizard captures.

ACOs were checked daily during pitfall trapping and mammal tracking sessions (see section 2.2.5 for details of mammal surveys). Any lizards flushed from cover while walking transects were also noted, although on most occasions they could not be identified. All lizards encountered were recorded, even if they could not be caught or identified. Note that a lizard did not have to be caught to be identified.

Lizard captures were too low to allow a meaningful species diversity index to be calculated, but the species diversity and total number of lizard encounters (from both the initial and main surveys, and from all survey methods) at each site was described. Only survey methods used across all sites (ACOs and pitfall traps) were used in comparisons between sites and statistical tests. To correct for unequal search effort between sites (due to time limitations), a relative abundance index was calculated by adding the standardised catch rate (catch per unit effort, CPUE) for each survey method at each site. All study sites were known to have native lizards present (confirmed by ACOs or hand searching), but at Lady Phoenix Reserve native lizards were only caught during hand searching. Because hand searching was not used across all sites it was not included in analyses, resulting in a relative abundance index of zero at Lady Phoenix Reserve. To correct for this, one capture was added to the ACO captures at all sites (table 2.2), which maintained the ranks of the relative abundance indices while correcting the erroneous zero. Based on this index, the six sites were classified as having relatively high, moderate or low abundance of native lizards. These classifications will be used in future chapters to investigate the effects of various habitat features on the abundance of native lizards. Average SVL, TVL and weight were calculated for each of the species caught during the main survey period, and the percentage of all species that had lost tails was calculated.

Comparing the efficiency of the different survey methods using an index such as catch per unit effort was not possible as ACOs and pitfall traps are passive methods, while hand searching is an active method and was not used at all sites. Therefore, the different survey methods were compared qualitatively, considering factors such as cost, ease of use, number of lizards positively identified and the amount of time required to install and check.

2.2.5 - Mammal surveys

At the conclusion of each pitfall trapping session, tracking cards baited with peanut butter were put out in tracking tunnels for one night before being removed (see figure 2.8 for schedule). This very short tracking period was used because during the initial surveys it was found that rats would locate more tunnels each night if baited cards were left out over consecutive nights, which would falsely inflate activity rates (see Appendix 1). Tracking cards were not run simultaneously with pitfall traps in order to avoid attracting rats to pitfall traps that may contain lizards, thereby increasing the predation risk. The percentage of cards tracked by each species was calculated as an index of activity at each site. In March 2013, after the three pitfall trapping sessions were completed, snap-trapping was undertaken to estimate the rodent density at each site. Snap trapping was left until the end of the study to avoid interference with activity rates recorded by tracking tunnels.

The percentage of cards tracked by each species was calculated at each site as an index of mammal abundance. A second index of mammal abundance was also calculated for each species at each site - the number of captures per 100 corrected trap nights (hereafter C/100TN = $n/(T-(0.5*I))*100$ where n = number of target captures, T = number of total trap nights, I = number of nights traps were unavailable). During each trap night some traps became inactive due to being tripped by rain or falling debris, or by catching an animal (including non-target species). The index used takes into account the number of nights that traps were unavailable by counting each of these instances as a half night only. Any instances where the bait was removed without the trap firing were not included in corrections, as these traps are still capable of catching a rodent (Gillies, 2013).

Chi squared tests were used to test whether the frequency of mammal and rat tracks and captures were distributed equally among the six sites. A significance level of 0.05 was used for all statistical tests unless otherwise specified. To assess the suitability of tracking tunnels as an alternative to snap traps for estimating rat abundance in urban forest, the relationship between tracking tunnel indices and snap trap catch rates was investigated using a Pearson correlation.

2.2.6 – Relationship between lizard and mammal abundance

The relationship between introduced mammals and native lizards was investigated using Spearman's rank correlation between rat abundance and activity and the relative abundance of native lizards. I hypothesized that sites with high abundance of introduced mammals would have lower abundance of native lizards, and vice versa.

2.2.7 – Survey techniques

Artificial Cover Objects (ACOs)

ACOs are a low-impact method that has been shown to be effective for surveys of both skinks and geckos (Lettink, 2007; D. J. Wilson *et al.*, 2007; Hoare *et al.*, 2009; Thierry *et al.*, 2009). ACOs consisted of 50 cm x 50 cm single sheets of Onduline (distributed by GBS Group), which retain heat and therefore are thermally attractive to reptiles. Three ACOs were installed at each sampling point (27 ACOs per grid) and where possible ACOs were placed near natural cover to increase the likelihood of use. Vegetation under ACOs was removed to aid in detecting and catching skinks (Lettink, 2007; Hoare *et al.*, 2009).

An ACO check involves one observer lifting the ACO while a second catches and identifies any lizards beneath it. . All ACO checks were conducted between 9:00 a.m. and 5:00 p.m. on clear, warm days in order to maximise the likelihood of capturing skinks (Hoare *et al.*, 2009) and the order that sites were visited in was varied.

Pitfall trapping

Three pitfall traps were installed at each sampling point (18 sampling points per site) within 0.5m of each other and each was randomly assigned to one of three bait types: banana, cat food, or control (no bait). These two baits were chosen in order to compare the effectiveness of fruit-based versus meat-based baits for omnivorous skinks such as copper and ornate skinks. Although pear is often used as a bait for lizards by the Department of Conservation (Hare, 2012b), banana was chosen for this study based on its success in attracting Moko skinks during surveys I was involved with on nearby Tiritiri Matangi Island. Cat food (the type sold in sausage-like rolls in the chilled department rather than the canned type) was chosen to be the meat-based bait in this study as it has a strong scent and is easy to cut into appropriately-sized pieces.

Pieces of bait approximately 1 cm^3 in size were used and baits were replaced as necessary. During pitfall trapping sessions traps were checked every 24 hours, and lizards caught in the traps were identified, weighed (to the nearest 0.5 g) and measured (snout-vent length, SVL and tail-vent length, TVL to the nearest millimetre). Tail loss and degree of regeneration was also noted as an indicator of predation pressure. All skinks were released near cover within 1 m of the point of capture. Traps were closed when not in use to prevent accidental captures.

Pitfall traps consisted of a flexible plastic sleeve 20 cm in height buried flush with the ground surface (plate 2.4 and 2.5), which prevented the earth from collapsing inward and crushing the plastic container inside. A 700 mL plastic container was placed in the bottom of each sleeve with a small piece of polystyrene to allow trapped lizards to float in the event of heavy rain causing the trap to flood. The second design included a second plastic container above the first with a plastic mesh bottom (gap diameter of 12 mm) to allow lizards to access the bait in the lower container and prevent entry of rodents which are well known to prey on lizards in traps (Newman, 1988, 1994; Towns & Elliott, 1996). This gap size should allow skinks to enter the traps but exclude rats. Geckos would also be excluded but the predation risk for trapped skinks was too great to justify removing the mesh. A 20 cm x 24 cm corflute lid was fixed above the trap using bamboo skewers to keep out the rain and also provide shade for trapped lizards. These traps were deployed at two sites (four grids) but after the first night of use it was found that the plastic mesh was not sufficient to exclude rats. Five traps in one grid had been chewed through by rats trying to get to the bait in the lower cup (plate 2.4).

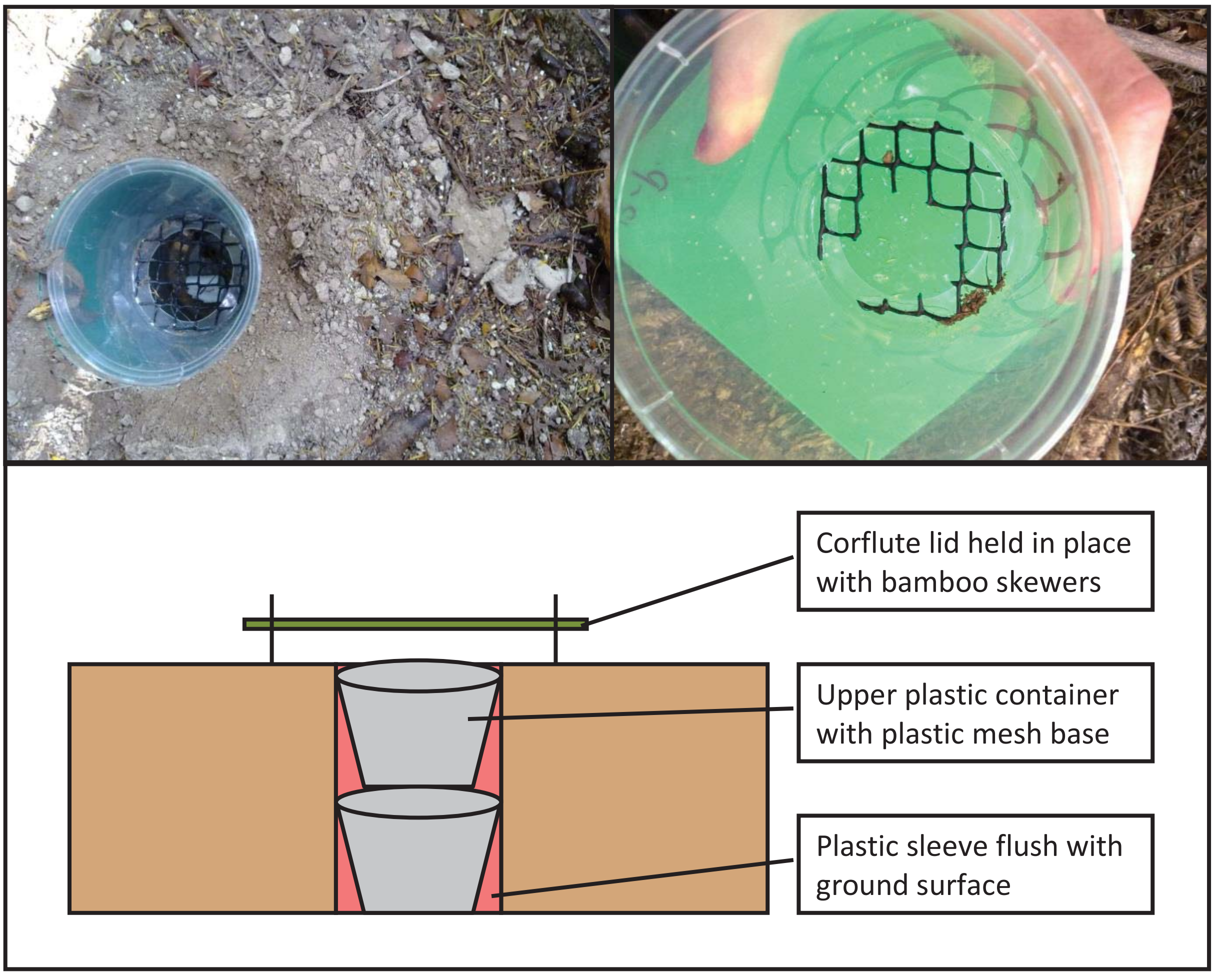

Plate 2.4 - Second pitfall trap design. Above left: trap in situ, the polystyrene float in the lower container can be seen through the mesh of the upper container. Rat droppings can also be seen on the right of the photograph. Above right: plastic mesh of upper container chewed through after first night of use. Below: diagram of second trap design.

In order to reduce the predation risk to any lizards caught in the traps the design was modified again, with the removal of the upper container and the top of the trap being covered by a square of steel mesh (gap size 12mm) fixed in place with metal pins (plate 2.5). Because the plastic containers fit snugly within the plastic sleeve, the containers could be placed just below the surface so that any lizards entering the trap would not fall so far.

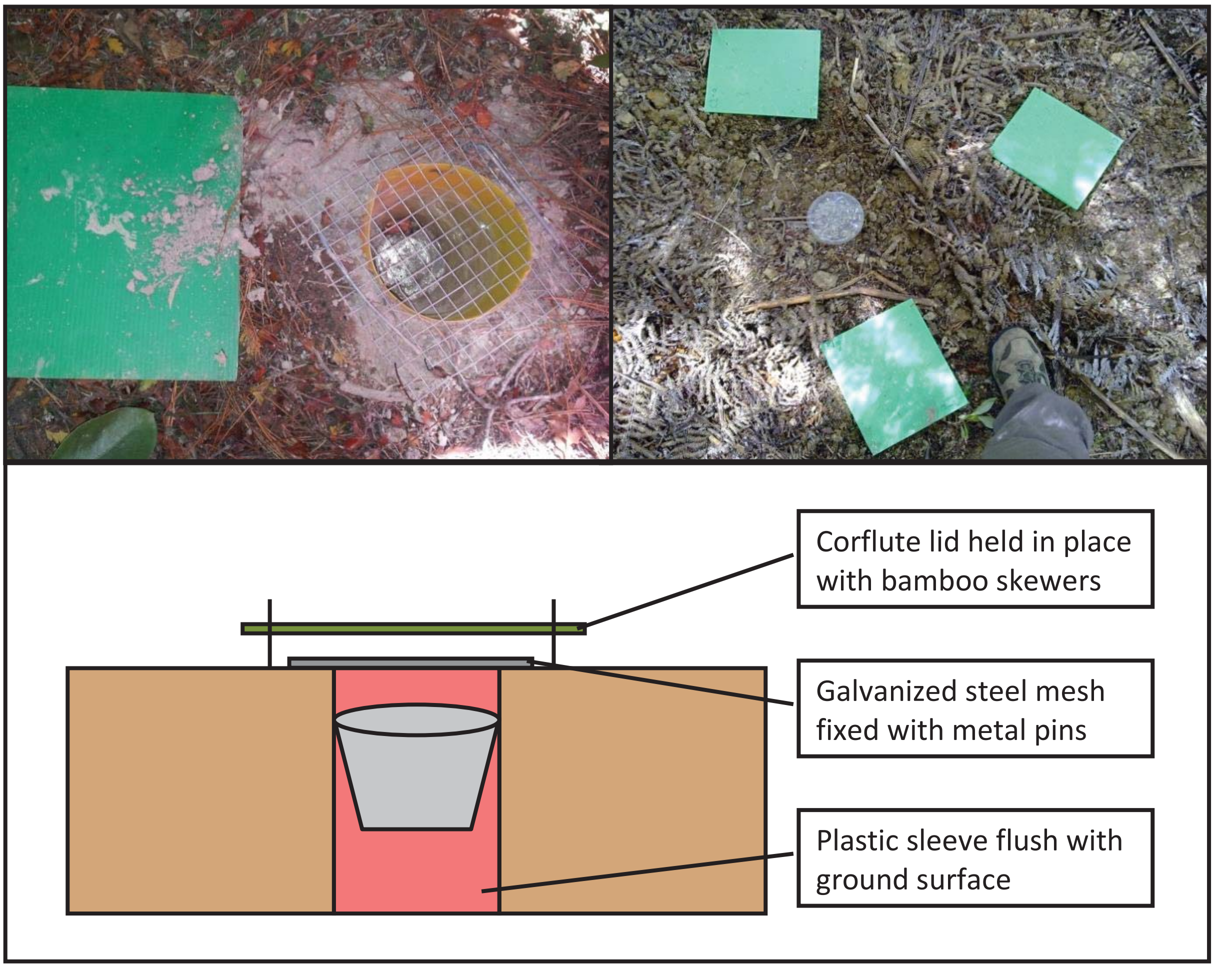

Plate 2.5 - Final pitfall trap design. Above left: An animal attempted to access this trap but was unable to move the steel mesh. Note the green corflute lid has been removed and loose soil from the animal digging can be seen on top of it. Above right: pitfall traps in situ. Below: diagram of final trap design.

Hand Searching

Hand searching involved searching within a three metre radius of each sampling point, particularly in and around leaf litter, fallen logs and thick vegetation where lizards may be located. The number of person-hours for each search was recorded (i.e. two people searching for 30 minutes equates to one person-hour). Because specific habitats are targeted, hand searching is a biased survey method and its use in this study was limited to confirming lizard presence. All hand searches were conducted between 9:00 a.m. and 5:00 p.m. on clear, warm days in order to maximise the likelihood of capturing skinks (Hoare *et al.*, 2009) and the order that sites were visited in was varied.

Hand searching was not carried out at Torbay Heights and Silverdale Scenic Reserve, where lizard presence had already been confirmed using ACOs. As this method was not implemented at all sites, lizard data from hand searching is presented for the total lizard encounters at each site but is not included in analysis

Tracking tunnels

Tracking tunnels (Black Trakka™) were used to detect mammalian predators. They were fixed in place with metal pins along the edge of banks, logs, tree roots and other areas that rats would be likely to use. Tracking cards were made up of shiny white cardboard strips, inked with non-toxic 'Black Track' ink in the centre (Pest Control Research Ltd.) and baited with peanut butter.

Snap trapping

One Victor® snap-trap was installed at each site and baited with peanut butter (plate 2.6). A square of chicken wire 50 cm x 30 cm in size was bent into a tunnel shape (leaving sufficient room for the bar of the snap trap to swing) and secured over each trap using metal pins in order to exclude non-target animals such as birds, possums (*Trichosurus vulpecula*) and cats. Traps were checked daily and re-baited as necessary for a total of 6 nights at each site (Dianne Brunton, personal communication). Animals caught in the traps were identified to genus, and rats were sexed and weighed. Although rats were the main target, other small mammalian predators such as mice and hedgehogs that were caught were also noted, as well as other non-target animals such as birds.

Plate 2.6 - Victor® rat traps used in this study. Note the yellow plastic treadle that fires the trap mechanism and where the bait is placed. On this occasion two mice (*Mus musculus*) were caught simultaneously.

2.3 Results

2.3.1 – Lizard abundance, diversity and morphometrics

A total of 4,698 ACO checks, 1,944 pitfall trap nights and 13 hours hand searching were conducted over the duration of the study, resulting in 85 lizard encounters (either seen or caught), only 32 of which could be identified (note that a lizard did not have to be captured to be identified) (table 2.1). Skinks classified as 'unidentified' were those that were not caught and therefore could not be positively identified. They were included in table 2.1 to give a picture of overall lizard activity at each site, but were not included in analyses as it was impossible to know whether multiple unidentified counters at a particular site represent different individuals or many sightings of one individual. The most common species encountered was the ornate skink, with 16 encounters (50 % of identified encounters).

Table 2.1 – Number of lizard encounters at each site over the entire study period. All survey methods are included. Numbers in brackets represent the number of individuals encountered while hand searching, which are not included in analyses.

Site	Copper skink	Ornate skink	Rainbow skink	Forest gecko	Unidentified skink	TOTAL
Eskdale	3 (3)	1	0	1 (1)	3 (1)	**8**
Kauri Glen	1 (1)	2	0	0	2	**5**
Kauri Park	1	11 (3)	0	0	5	**17**
Lady Phoenix	1 (1)	0	0	0	1	**2**
Silverdale Scenic	1	0	2	0	8	**11**
Torbay Heights	0	2	6	0	34	**42**
TOTAL	**7**	**16**	**8**	**1**	**53**	**85**

Torbay Heights had the highest number of encounters, with 42 of the 85 lizard encounters. However, it should be noted that only eight of the 42 skinks that were encountered at Torbay Heights could be identified (see 2.4.1). The majority of skink encounters at Torbay Heights (74 %) occurred at just four of the 18 sampling points, where skinks were frequently disturbed while basking. Kauri Park had the second highest number of encounters with 17 encounters, but in contrast to Torbay Heights only five of these were not able to be identified. Lady Phoenix had the lowest encounter rate, with just two skinks encountered here during the study. In terms of lizard diversity, Eskdale Reserve had the most species, with three native species found at this site (copper and ornate skinks and forest gecko). Kauri Park and Kauri Glen had two native skink species (copper and ornate), and Silverdale scenic Reserve and Torbay Heights had one native and one introduced species each. Lady Phoenix Reserve had the lowest diversity with only one species identified.

Interestingly, a single forest gecko was found on the ground in leaf litter during the day while hand searching at Eskdale Reserve, which is unusual considering this species is arboreal and mainly nocturnal (plate 2.7). It should be noted that this individual had recently shed its tail and had several superficial abrasions where scales were missing, potentially due to an encounter with a predator or an aggressive event with a conspecific.

Plate 2.7 - Left and middle: the single forest gecko (*Mokopirirakau granulatus*) found during lizard surveys (photos by Luca Butikofer). Right: One of the sloughed skins found at the same site as the forest gecko.

Two sloughed gecko skins were found on top of ACOS three months after the gecko capture and are likely to be from forest geckos based on the pattern. Both skins were found on the same day and within approximately 100m of each other. As the site was surveyed the previous day and no skins were found, and due to their delicate nature it is likely they had been shed very recently (within 24 hours), suggesting at least two individuals were present. As the animals were not caught I cannot confirm that one of the sloughed skins was not from the same individual caught three months earlier and therefore these were excluded from analysis.

Kauri Park had the highest relative abundance of native lizards and was classified as such (table 2.2). Kauri Glen and Eskdale Reserve and Torbay Heights were classified as having moderate lizard abundance, and Lady Phoenix and Silverdale Scenic Reserves were considered to have only low abundance of native lizards.

Table 2.2 - Relative abundance indices of native skinks at each site. CPUE = catch per unit effort. * = corrected for sites which had no ACO captures but lizard presence was confirmed by hand searching, see section 2.2.3.

Site	Corr ACO captures*	Effort (checks)	CPUE	Pitfall captures	Effort (trap nights)	CPUE	Relative abundance index	Relative abundance category
Eskdale	3	702	0.0043	0	324	0.0000	0.0043	Moderate
Kauri Glen	3	702	0.0043	0	324	0.0000	0.0043	Moderate
Kauri Park	9	864	0.0104	1	324	0.0031	0.0135	High
Lady Phoenix	1	702	0.0014	0	324	0.0000	0.0014	Low
Silverdale Scenic	2	864	0.0023	0	324	0.0000	0.0023	Low
Torbay Heights	3	864	0.0035	0	324	0.0000	0.0035	Moderate
Total	16	4698	0.0034	1	1944	0.0005	-	-

In total, 22 skinks were captured (using all methods) during the study period, nine of which had regenerated tails (41 %). Captured skinks were not weighed and measured in the initial survey, therefore morphometric measurements are only available for the nine individuals caught during the main survey period (table 2.3). One of the nine skinks was a neonate rainbow skink, which was excluded when calculating averages. A total of five ornate skinks were captured in the main surveys, with average measurements as follows: SVL = 48 mm, TVL = 35 mm, weight = 3.2 g. As is to be expected, the three adult rainbow skinks had smaller average measurements: SVL = 44 mm, TVL = 64 mm, weight = 2.0 g. Note that although the average SVL is quite similar between the two species in this sample, the average tail length of rainbow skinks is almost twice as long as that of the ornate skink.

Table 2.3 - Morphometrics of skinks captured during the main survey January to April 2013. SVL = snout-vent length. TVL = tail vent length. ** = this individual was a neonate and was not weighed. Tail loss = any portion of the tail has been lost, whether fully regenerated or not.

Species	Site	Method	SVL (mm)	TVL (mm)	Weight (g)	Tail loss
Ornate	Eskdale	ACO	58	32	4.0	Y
Ornate	Kauri Glen	ACO	42	14	2.0	Y
Ornate	Kauri Glen	ACO	44	40	3.0	N
Ornate	Kauri Park	Trap	56	53	4.5	N
Ornate	Kauri Park	ACO	42	36	2.5	Y
Rainbow	Silverdale Scenic	ACO	41	67	2.0	N
Rainbow	Silverdale Scenic	Flush	21	30	**	Y
Rainbow	Torbay Heights	ACO	44	41	2.0	Y
Rainbow	Torbay Heights	Trap	47	83	2.0	N

2.3.2 – Comparison of lizard survey methods

The largest numbers of encounters were from ACOs and flushing, which both had similar numbers of encounter. ACOs allowed more lizards to be identified – 41 % for ACOs compared with 17 % for flushing (table 2.4). A large proportion of skinks using ACOs could not be identified, either because they fled as the observers approached (13 of 22 unidentified encounters, 35 %) or evaded capture during ACO checks (9 of 22 unidentified encounters, 24 %). Of the six lizards identified after being flushed, only one was caught (table 2.3).

Table 2.4 – Number of lizards encountered using four different survey methods over the entire study period. For descriptions of each method, see sections 2.2.2 and 2.2.3.

Method	Copper skink	Ornate skink	Rainbow skink	Forest gecko	Unidentified skink	TOTAL
ACO	2	11	2	0	22	**37**
Hand search	5	3	0	1	1	**10**
Pitfall trap	0	1	1	0	0	**2**
Flushed	0	1	5	0	30	**36**
TOTAL	**7**	**16**	**8**	**1**	**53**	**85**

Although hand searching was not undertaken at every site, it appears to be an efficient method in urban forest fragments, resulting in 10 encounters from 13 hours of searching (table 2.5). Some sites were particularly successful, for example, in just one two-hour long session at Kauri Park, three skinks were caught. Other sites had more variable hand searching success, such as Eskdale reserve. A total of four hours searching over two sessions had been undertaken and resulted in no lizards being caught, until a third session of three hours was carried out, during which 5 lizards were found.

Table 2.5 - Number of lizards captured and key benefits of each survey method.

Method	Lizards caught	Effort	Efficiency	Cost per unit	Easy to install	Time to install	Predation risk	Observer skill	Time per sampling point
ACO	37	4,698 checks	Low	High	Yes	Low	No	High	Low
Hand search	10	13 hours	High	n/a	n/a	n/a	n/a	High	High
Pitfall trap	2	1944 nights	Low	Moderate	No	High	Yes	Low	Moderate

Pitfall trapping had the lowest capture rate and was by far the least efficient method employed, with a total of 1,944 trap nights resulting in just two captures, and as a result the influence of bait type on captures was not able to be analysed. In this study, ACOs also had low efficiency, resulting in 37 captures from 4,698 ACO checks, which equates to one capture every 127 ACO checks.

2.3.3 – Mammal abundance and diversity

A total of 324 tracking nights were carried out during the main survey period, resulting in 123 predator tracks being recorded. Chi squared tests indicated that both the number of all mammal tracks ($\chi^2 = 41.63$, df = 5, $p < 0.001$) and the number of rat tracks ($\chi^2 = 82.68$, df = 5, $p < 0.001$) were significantly different between sites. Torbay Heights had the highest mammalian predator activity with 72.2 % of tunnels recording tracks over 54 tracking nights, and the majority of tracks being from rats (table 2.6). Kauri Park also had high rat activity, with 53.7 % of tunnels recording rat activity. Eskdale Reserve on the other hand had a very low mammal activity with just two tracks recorded, which equates to less than 4 % of tunnels showing any mammal activity.

Overall, rat tracks were by far the most common, accounting for more than half of the total number of recorded tracks (75 of 123 tracks). Lady Phoenix and Silverdale Scenic Reserves were the most diverse in terms of mammal tracks, having mice, rats and hedgehogs present. Rats were present at all sites, hedgehogs were at all sites except Torbay Heights, and mice were only present at three of the six sites and had low tracking rates. In addition to the mammalian predator tracks, on 3 occasions lizard tracks were found in tracking tunnels, and on 28 occasions possum tracks were found but they are not included here as the size, location and bait choice used in tracking tunnels may have produced a biased sample.

Table 2.6 – Mammal activity indices - percentage of tracking cards tracked by predatory mammals over the main survey period.

	Hedgehog	Mouse	Rat	TOTAL (%)
Eskdale	1.9	0.0	1.9	**3.7**
Kauri Glen	29.6	0.0	1.9	**31.5**
Kauri Park	9.3	0.0	44.4	**53.7**
Lady Phoenix	24.1	3.7	16.7	**44.4**
Silverdale Scenic	3.7	11.1	7.4	**22.2**
Torbay Heights	0.0	5.6	66.7	**72.2**

Over 567 trap nights, a total of 74 mammalian predators were caught across the six sites (table 2.7). Chi squared test showed that the total number of mammal ($\chi^2 = 17.14$, df = 5, $p < 0.005$) and rat ($\chi^2 = 32.65$, $p < 0.001$) captures was significantly different between the sites. As with the activity rates, Torbay Heights also had the highest mammal abundance index, with 24.4 C/100TN, and the highest total capture of rats. Conversely, Eskdale Reserve had a very low number of rats caught, and also a very low mammal abundance index. Silverdale Scenic was unusual in that just one rat was caught, and the numbers of mice caught was high when compared with the other sites. It should be noted, however, that the traps used were rat traps and they are not optimized for catching mice. Hedgehogs were also caught infrequently and again, the traps were not designed for this species. As such, the relative abundance values for both mice and hedgehogs should be treated with caution, as the capture rates of these species may be unreliable.

However, the patterns in species abundance values for mice and hedgehogs across the study sites are similar to those of the activity indexes, which are calculated using data from tracking tunnels which are suitably sized for all three species. Rats were by far the most abundant small mammal over all sites, with a relative abundance index of 9 C/100TN, compared with 1.6 C/100TN for hedgehogs and 2.5 C/100TN for mice.

Table 2.7 – Counts of predatory mammals caught at each site during kill trapping. * = corrected for the number of nights traps were unavailable, see section 2.2.4. C/100TN = captures per 100 trap nights.

Site	Hedgehog	Mouse	Rat	TOTAL MAMMAL CAPTURES	Corrected trap nights*	Mammal abundance index (C/100TN)
Eskdale	0	1	2	**3**	102.5	**2.9**
Kauri Glen	6	0	4	**10**	95.5	**10.5**
Kauri Park	0	0	11	**11**	96	**11.5**
Lady Phoenix	3	1	13	**17**	93.5	**18.2**
Silverdale Scenic	0	10	1	**11**	89.5	**12.3**
Torbay Heights	0	2	20	**22**	90	**24.4**
TOTAL	**9**	**14**	**51**	**74**	**567**	-
Species abundance index (C/100TN)	**1.6**	**2.5**	**9.0**	-	-	-

2.3.4 – Comparison of rat survey methods

Both the tracking card and kill trapping data showed similar patterns of rat abundance between sites, with Torbay Heights having an abundant rat population, Kauri Park and Lady Phoenix Reserve having moderate rat abundance, and the three remaining sites having only low numbers of rats (tables 2.7 and 2.8). A Pearson correlation showed a positive correlation between the percentage of cards tracked by rats and the number of rats caught per 100 trap nights, with the percentage of cards tracked accounting for 79.6 % of the variability in C/100TN (r = 0.892, p = 0.009).

2.3.5 – Relationship between lizard and mammal abundance

Spearman's correlations did not reveal any significant correlations between the relative abundance of native lizards captured and mammal abundance, rat abundance or rat activity (table 2.9).

Table 2.8 – Spearman's correlations between the relative abundance of native lizards and various measures of small mammal abundance at all sites. Significant correlations are shaded, α = 0.05.

	r	*p*
mammal relative abundance (C/100TN)	-.638	.173
rat relative abundance (C/100TN)	-.116	.827
rat activity (% of cards tracked)	-.118	.824

2.4 – Discussion

2.4.1 – Lizard abundance, diversity and morphometrics

To examine the factors that may contribute to the coexistence of lizards and rats in urban forest fragments and to evaluate the success of any management attempts, effective methods of estimating the abundance of both species must be identified. The high variability in lizard encounters between sites, survey methods, and lizard species could significantly bias overall abundance estimates if not accounted for. For example, the lizard encounter rate was unusually high at Torbay Heights, but the capture rate did not reflect this, as only 3 lizards were caught here during the study. The high encounter rate at this site is mainly due to four sampling points which accounted for 31 of the 42 encounters at Torbay Heights. Three of these points were near the top of a ridge where the vegetation changed to open scrubland with few trees, consisting mainly of low, dense vegetation such as toetoe (*Austroderia* spp) and tangle fern (*Gleichenia dicarpa*). One point was directly adjacent to a large fallen pine tree, which provided abundant retreat sites among the bark and debris, and the large canopy gap it caused meant that direct sunlight for basking was freely available. The abundance of natural basking and retreat sites at these points may mean that ACOs were not as attractive and may explain why skinks were never found under ACOs at this point. Additionally, skinks were able to maintain thermally optimal body temperatures even in the early morning and were capable of moving quickly when disturbed (Hoare *et al.*, 2009), retreating into one of the many available refuges and therefore I was unable to opportunistically catch any skinks at this point.

Rainbow skinks were captured and identified at two of the four sampling points which together accounted for 31 of the 42 encounters at Torbay Heights, and based on the relatively open habitat at all four points, the small body sizes and comparatively long tails of skinks observed (table 2.3), it is likely that these 31 unidentified encounters were rainbow skinks. Rainbow skinks are fairly robust to human disturbance (Peace, 2004) and are therefore less likely to be displaced by repeat sampling and on one occasion three individual skinks were seen at one sampling point, suggesting that these 31 encounters in fact represent multiple encounters of a small number rainbow skinks at each point. Similarly, six of the eight unidentified encounters at Silverdale Scenic Reserve are also likely to represent multiple sightings of one individual rainbow skink. On 5 separate occasions, a skink of similar size and colouration was seen at the same sampling point, sometimes under an ACO, but would flee as I approached and I was unable to catch it. On the sixth occasion, a skink was finally caught and identified as a rainbow skink, and assuming the previous sightings were of the same individual this would reduce the unidentified encounters at Silverdale Scenic Reserve to two. Given the high likelihood that many of the unidentified skink encounters at Torbay Heights and Silverdale Scenic Reserve were rainbow skinks, Kauri Park probably has the greatest abundance of native skink species. This is despite the moderate density of rats at this site (table 2.7 and 2.8) and therefore the habitat features here may be particularly interesting in terms of coexistence with introduced rats (see chapter 3). It should be noted that this study did not examine whether these lizard populations are viable, and therefore even the most ‘abundant’ population may be declining.

One of the limiting factors in analysing lizard abundance and diversity between sites was the fairly small sample size of positively identified skinks. This was partly due to the fact that a large portion of the skinks encountered in this study were flushed from cover while observers walked transects checking ACOs and pitfall traps, and as such they couldn't be caught or identified. Flushing may have been more of a problem in this study as it was conducted in forest, where most other published studies have been done in grassland. The act of moving through forest inevitably involves disturbing vegetation and fallen debris which creates noise and vibration that could causes skinks to flee ACOs more readily than in open grassland. Another major factor contributing to the small sample size of identified skinks was the very low efficiency of pitfall traps and ACOs used in this study (section 2.4.2). The number of skink captures may have been improved with more extensive hand searching, but due to a lack of experienced researchers and strict time limitations this was not possible.

Tail loss was fairly high among the lizards captured in this study (41%), either due to high levels of aggression between skinks, or frequent predation attempts (Towns, 1975; Patterson, 1992; Hayes *et al.*, 2012). Given the relatively low encounter rates and abundances of skinks at most sites, it is unlikely that aggression levels between skinks is very high and is more likely that the high rate of tail loss observed is due to predation events (Hayes *et al.*, 2012). An important source of predation attempts on skinks is likely to come from rats, but also from feral and pet cats roaming in urban bush fragments, which were not surveyed for several practical reasons (see section 2.4.3).

2.4.2 – Comparison of lizard survey methods

The use of ACOs in urban forest fragments

ACOs were chosen for use in this study as the literature suggests that they are relatively inexpensive, easy to install and provide accurate and precise indices of abundance of cryptic herpetofauna. Additionally, ACOs carry with them lower predation risks for lizards using them as lizards are free to enter and leave and ACOs less affected by observer bias than other methods such as hand searching. (Hoare *et al.*, 2009; Lettink *et al.*, 2011; Anderson *et al.*, 2012). However, the efficiency of ACOs in this study was very low, equating to 0.003 CPUE (catch per unit effort), or one capture per 333 checks. Other studies on native skink species have reported captures equating to CPUE values of 0.074 to 0.653 (table 2.10).

Table 2.9 – Number of captures and ACO checks reported by studies on native skink populations in New Zealand. Catch per unit effort (CPUE) = captures/effort (ACO checks).

Study	Captures	ACO checks	CPUE
This study	15	4698	0.003
Lettink (2007)	160	2160	0.074
Hoare *et al.* (2009)	1175	1800	0.653
Lettink *et al.* (2011)	866	1400	0.619

There are several explanations which may have contributed to the unusually low capture efficiencies observed in this study compared with other published studies. Firstly, the low efficiencies may simply be a reflection of low population densities in the urban forest fragments used in this study. It is likely that the density of lizards in this type of habitat is lower than was seen in other published studies, however the number of lizards seen while walking transects in this study suggests that lizards are more abundant than the methods used indicate, and that there are likely to be other factors contributing to the low efficiencies seen.

One of these factors is that most previous studies were conducted in grassland, duneland or coastal scrub habitats, which are significantly more open and typically have little canopy cover compared with the forest habitats in this study. Additionally, many of these published studies were undertaken in the southern North Island and throughout the South Island, where temperatures are significantly cooler than the Auckland region where this study was located. The combination of warmer environmental temperatures and more canopy cover with reduced solar penetration may combine to result in ACOs in this study being less thermally attractive to skinks and therefore resulting in lower capture rates than were seen in previous studies (Hoare *et al.*, 2009). Grassland and scrub-type habitats may also provide fewer natural refuges when compared with native forest, which often contains deep leaf litter, low-growing vegetation, and fallen logs and branches which lizards may use as natural refuges. An abundance of natural refuges combined with the thermal attributes of the habitat may make ACOs less attractive to skinks and therefore less efficient in forest environments.

A second potential explanation for the lower efficiencies of ACOs seen in this study is the prevailing environmental conditions during the main survey period. During the summer of 2012 – 2013, much of New Zealand experienced warmer temperatures, more sunshine, and lower rainfall than normal, leading to a drought being officially declared in several regions including Auckland, which had one of the worst droughts in more than 40 years (Porteous & Mullan, 2013). Native skinks are known to be susceptible to cutaneous water loss (Neilson,

2002) and have been shown to be less active when ambient temperature is above 18°C (Hoare *et al.*, 2009), and as such they may be adversely affected by drought conditions. In order to minimize evaporative water loss in such warm, dry conditions, skinks may have been forced to limit their activity and take refuge in more damp areas, therefore would be less likely to encounter and use ACOs.

Another contributing factor to the low efficiency of ACOs in this study is the incidence of skinks fleeing ACOs. Of the 22 unidentified skinks using ACOs, the majority fled as observers approached and could not be caught. Observers always approached ACOs as quietly as possible, but even so, skinks may see, hear or feel observers approaching and flee before observers are close enough to be able to catch them. In a study in Otago, it was shown that skink recaptures are generally low compared with geckos, suggesting a lower tolerance to disturbance (D. J. Wilson *et al.*, 2007). Even more importantly, the same study showed that captures of skinks declined when ACO checks were carried out daily, but not when they were carried out weekly, giving further support to the idea that disturbance frequency may affect skink captures. This may lead to reduced use of ACOs over time or an increased likelihood of skinks fleeing upon observers approaching. In this study, ACO's were checked every 3 – 5 days during the initial surveys. In the main survey period ACO's were checked daily during each sampling session, which lasted three days, and then were undisturbed for a minimum of 14 days until the next sampling session. However, there did not appear to be a difference in the capture frequency of skinks between the two sampling regimes in this study. Skinks that evaded capture usually ran into nearby cover. This is unavoidable, as ACOs nearer cover are more successful than those further from cover (Lettink, 2007), and the thermal properties of ACOs mean that skinks under them are often warm and capable of running quickly (Thierry *et al.*, 2009). Ideally two observers are present when checking ACOs – one lifts the ACO and the other catches any skinks beneath.

ACOs had the highest equipment cost per unit (approximately $5.00 NZD), and a moderate to high level of observer skill is required to prevent lizards from escaping without causing injury. However, there is no increased predation risk to lizards and they are very fast and easy to check and install. It should be noted that although the time taken to install the ACOs themselves is very low, the grids that the ACOs and pitfall traps were located on were often time-consuming to set up due to encountering impassable features such as cliffs and streams, or extending beyond the boundaries of the forest fragments.

The use of pitfall traps in urban forest fragments

The efficiency of pitfall traps in this study was extremely low at 0.001 CPUE, equating to one capture per 1000 trap nights. Other studies of native skink species have reported captures equating to CPUE of 0.112 to 0.171 CPUE (table 2.11). As only two skinks were captured, the efficiency of the different bait types used could not be assessed.

Table 2.10 - Number of captures and pitfall trap nights reported by studies on native skink populations in New Zealand. Catch per unit effort (CPUE) = captures/effort (trap nights).

Study	Captures	Trap nights	CPUE
This study	2	1944	0.001
Towns and Elliott (1996)	2897	23667	0.122
Lettink (2007)	329	1920	0.171
Lettink and Seddon (2007)	536	4800	0.112
Lettink *et al.* (2011)	542	3400	0.159

The drought conditions during the summer of 2012-2013 may have contributed to the low captures of pitfall traps in this study for similar reasons as discussed above. Pitfall trap design may also partially explain the low capture rates observed. Because the traps were being installed in areas were predators were known to be present, extra precautions were implemented in order to reduce mortality of trapped lizards. The mesh that was put across the top of the pitfall traps in order to exclude predators may have discouraged skinks from entering traps. If pitfall traps are to be used in future studies in areas where predators are known to be present, the use of mesh over the top of pitfall traps must be carefully considered in terms of the level of predation risk to trapped lizards, and if the predation risk is deemed to be too great, then other methods such as hand searching or ACOs should be considered.

Although the cost per unit (approximately $3.00 NZD) and observer skill level required for pitfall trapping is relatively low, they were very difficult and time consuming to install (table 2.5). The traps are made up of several parts which must be assembled in situ, and digging over 300 holes in compacted soil among tree roots proved so problematic that a petrol-powered borer operated by two strong men was required, and even then was extremely difficult.

Comparing lizard survey methods

While using a diverse range of survey methods is often recommended in order to maximise detection (Anderson *et al.*, 2012), in practice a single method that is both time- and cost- efficient is usually preferred. Considering the efficiency of captures, time involved in setting up and overall cost of survey equipment the results of this study suggest that hand searching may be the most appropriate method for surveying relatively low-density lizard populations in forests with predators present. This method carries with it several limitations that do not apply to ACOs or pitfall traps used in a grid system. Firstly, hand searching is a biased survey method as it targets specific habitats (although a grid system may be used to reduce this), and to some degree requires previous knowledge of a species' habitat preferences. This method is also more susceptible to biases due to the skill of observers involved, as they must detect and capture lizards. However, if used by skilled observers, the advantages of hand searching over ACOs and pitfall traps suggest that it may be the most appropriate survey method for urban forest fragments. Advantages of hand searching are that it does not involve any equipment costs, time to install, or increased predation risk to lizards. However, it does take a relatively large amount of time to search each individual sampling point and a high level of observer skill is required.

Comparison of the efficiency of the lizard survey methods used in this study posed several problems, the first being that the unit of effort was different for every method used and therefore CPUE cannot be directly compared between different methods. Pitfall trapping, for example, is measured in trap nights (24 hour periods), hand searching is measured in person hours, and ACOs are measured by the number of checks. Pitfall trapping is a cumulative method, with lizards being able to be caught and detected over the entire 24 hour period that they are set. ACOs, on the other hand, only provide a very limited snapshot of the use of retreats at the time that they are checked. A second issue with using CPUE to compare methods is that some methods required considerable time in setting up, while other methods had little to no set up time required, and this is not taken into account with standard CPUE calculations. For example, the least efficient method employed in this study, pitfall trapping, would have been even less efficient if the many hours spent building traps and digging the holes were included in the calculation of CPUE. Hand searching, however, requires no set up time and resulted in the highest number of catches per unit effort.

2.4.3 – Mammal abundance and diversity

The abundance of rats in some urban forest fragments in this study was very high – 24.4 C/100TN at Torbay heights and 18.2 C/100TN at Lady Phoenix Reserve. These are much higher than the maximum C/100TN reported by some studies in large tracts of forest (both native and exotic) in rural areas, for example, King *et al.* (1996b) reported captures of rats up to 9.18 C/100TN, and (Efford *et al.*, 2006) reported captures up to 7.5 C/100TN. These results suggest that rats may be able to maintain very high densities in urban forest fragments, possibly as a result of anthropogenic food sources in adjacent urban areas, and therefore the predation pressure they exert on urban skink populations may be comparatively greater than in rural forests.

Cats were not surveyed due to the significant ethical issues with trapping pets, and their exclusion from surveys of mammalian predators is an important limitation of this study. The use of camera traps is one method that does not carry these ethical concerns but was not financially viable for this study. The mean home range size of cats in suburban New Zealand has been estimated to be 1.5 ha – 3.2 ha (depending on the method used) (van Heezik *et al.*, 2010). Based on these home range size estimates, cats almost certainly visited all sites, due to the proximity of suburban homes, and evidence of cat visits was seen throughout the sampling period, not just when pitfall traps were baited with cat food (pers. obs.). On one occasion a feral cat was seen at Kauri Park, and pet cats were seen at Kauri Park, Kauri Glen, Torbay Heights and Lady Phoenix Reserve (see plate 2.8).

Plate 2.8 - One of the pet cats (*Felis catus*) regularly seen visiting sampling points at Kauri Glen. Here it is seen within a few metres of one of the ACOs.

The impact of cats on wildlife populations can be serious, and cats are well known to be capable of exterminating entire populations (Iverson, 1978; Gillies & Clout, 2003; Galbreath & Brown, 2004; Medina *et al.*, 2011). One example comes from a 22 year long study in a suburban part of Australia: a single pet cat was thought to be responsible for the extinction of a local population of a rare lizard in just two years, and the population did not begin to recover until six years after the cat left the area (Bamford & Calver, 2012). Cats are also known to be important predators of lizard populations in New Zealand, where lizards have been found to be the third most important prey group for cats in suburban Auckland (Gillies

& Clout, 2003). As such, it is important to consider the effects of cat predation on urban lizard populations. Although trapping is not a feasible method of assessing cat activity, the results of tracking cards for estimating rat abundance in this study suggest that tracking of predatory mammals may provide a low-cost, low-impact alternative to trapping (section 2.3.4), and may be modified for surveying cat populations.

2.4.4 – Comparison of small mammal survey methods

The exclusive use of kill-trapping would have provided a very limited view of the distribution and activity of rats and other small mammals because individuals are removed from the population as they are sampled. To avoid this, a longer period of tracking was carried out before kill-trapping was conducted, however there are some limitations to consider with the use of tracking tunnels for small mammal surveys. Firstly, the use of tracking tunnels does not allow the number of individuals visiting the tracking tunnel to be determined, and therefore any one track may be from a single animal, or from several. Likewise, tracks in adjacent tunnels may be due a different individual visiting each tunnel, or one individual visiting several tunnels. Studies have shown that tracking rates provide fairly accurate measures of relative abundance when compared with more conventional data from kill trapping (K. P. Brown *et al.*, 1996), but for the reasons above tracking rates should be interpreted with caution. As discussed in section 2.3.4, the patterns of rat abundance were fairly similar between the two methods in this study, suggesting that the use of tracking cards to estimate rat abundance is appropriate in urban bush fragments. Secondly, some studies have found evidence that mice can be excluded from using tracking tunnels in the presence of rats, although the reasons for this are not clear (K. P. Brown *et al.*, 1996; Blackwell *et al.*, 2002). As such, tracking rates for mice in this study should also be interpreted with care as they may underestimate abundance.

Although it is often the preferred method for density estimates, kill trapping also has several limitations which must be considered. Firstly, trap rates for non-target species may not be reliable. The trap rates for mice are likely to be inaccurate because they are significantly lighter than rats and may be able to visit traps without setting them off, and are small enough to avoid capture on some traps that do fire (Watkins *et al.*, 2009, 2010). Hedgehog trap rates may also be inaccurate due to the covers that were put in place in an effort to exclude non-target species. However, some individuals were still able to access the traps. Additionally, previous surveys in nearby reserves indicate that cat food is strongly preferred over peanut butter by hedgehogs (author, unpublished data), and as peanut butter is not an optimal bait for attracting hedgehogs, it is possible that only a subset of the population was sampled.

The presence of non-target species can also be a problem when kill-trapping because they can result in high numbers of unavailable trap nights. At Silverdale Scenic Reserve, for example, the number of unavailable trap nights was unusually high (with traps being unavailable on more than 40% of nights they were set). This may be due to interference by possums which had by far the highest tracking rate at this site (96 % of all possum tracks are from this site), and are large enough to set off traps without being caught. Mice may also have contributed to the unusually high number of unavailable trap nights at Silverdale

Scenic Reserve, both by setting off traps without being caught, and by making traps unavailable for catching rats when mice were caught (Watkins *et al.*, 2009, 2010).

2.4.5 – Relationship between lizard and mammal abundance

No significant association between the relative abundance of small mammals and lizards was identified in this study. One possible explanation for this is that the relationship between predator and native lizard abundance may be non-linear, with predators having greater effects on low-density lizard populations than they do on populations of higher densities (van Heezik & Ludwig, 2012). Another potential explanation for this relates to sampling problems such as inefficiency of the survey methods used leading to potentially inaccurate lizard relative abundances. The apparent lack of association between the relative abundance of small mammals and lizards may also be due tothe habitat structure of urban bush fragments allowing lizards to escape or avoid rats. If lizards were able to escape and take refuge from predators safely, they may be able to coexist even where rats are present in high densities (although the lizard survey methods used in this study would be insufficient to detect this). The habitat structure of the sites will be examined in chapter 3. A second potential explanation is that lizards may have developed behaviours that help them avoid rats. Duvaucel's geckos are known to avoid microhabitats used by rats, although their habitat use does overlap on a larger scale (Hoare *et al.*, 2007b). This may also be the case for skinks in urban forest fragments, but it was not investigated in this study.

Chapter 3

Habitat parameters and the abundance of ground-dwelling lizards in urban forest fragments

Plate 3.1 - Forest canopy dominated by silver fern (*Cyathea dealbata*) and Tanekaha/Celery Pine (*Phyllocladus trichomanoides*).

3.1 – Introduction

3.1.1 – Habitat influences on lizard distribution

Habitat features are an important driver of spatial distribution in reptiles, both through habitat availability for, and habitat selectivity of, individual species (Johnson *et al.*, 2006; Garda *et al.*, 2013; McDonald & Luck, 2013; Stellatelli *et al.*, 2013). One of the key reasons that habitat is such an important factor in determining lizard distribution is its influence on abiotic factors. Lizards are ectothermic, meaning that physiological mechanisms play little or no part in the regulation of their body temperatures. Instead, many lizard species use behavioural adjustments in order to maintain their body temperatures, taking advantage of environmental heat sources or sheltering from the heat as necessary (Huey & Slatkin, 1976). Environmental temperature and solar penetration are two crucial factors for thermoregulation in reptiles, and can be significantly influenced by habitat features such as canopy openness, vegetation density, elevation and aspect (Huey & Slatkin, 1976; Stevenson, 1985). Some lizard species preferentially use particular habitats that provide sufficient opportunities for basking or avoiding temperature extremes, and others avoid thermally suboptimal habitats, both of which can have pronounced impacts on species distribution (Stevenson, 1985; Vitt *et al.*, 1997; Sartorius *et al.*, 1999; Stellatelli *et al.*, 2013). Human-induced alteration of habitats can influence lizard thermoregulation and distribution. For example, the introduction of exotic *Acacia longifolia* trees to native coastal grasslands in Argentina led to increased shading and lower temperatures under the introduced trees, which has reduced the thermoregulatory efficiency of the native lizard

Liolaemus wiegmannii. Another native species, *L. multimaculatus*, responded to the introduction of the exotic trees by restricting its activity to open habitats that resemble the native grassland, thereby retaining thermoregulatory efficiency but changing its distribution on a microhabitat scale (Stellatelli *et al.*, 2013).

The effect of habitat on thermoregulation is also a driver of temporal differences in lizard distribution, both on seasonal and daily scales. Some lizard species have been observed to utilize specific habitats in different seasons to optimize thermal efficiency, either by using warmer habitats in the cool season, or cooler habitats in the warm season (Christian *et al.*, 1983; Kerr *et al.*, 2003). On a daily scale, many diurnal lizards move to more open habitats or microhabitats in the early morning to maximise thermoregulation, and return to more shaded habitats to avoid overheating later in the day (Gifford *et al.*, 2012; Pelegrin *et al.*, 2013). Thermoregulation is one of the most important physiological processes in reptiles, having effects on activity, reproduction, and metabolism (Angilletta *et al.*, 2002; Gifford *et al.*, 2012; Monasterio *et al.*, 2013). Thermoregulation also carries with it costs such as energy expenditure when moving between basking and refuging microsites, exposure to predators when basking, and time that could otherwise be used for feeding or breeding (Huey & Slatkin, 1976; Downes, 2001; Gvozdik, 2002), and as such selecting habitats with optimal thermal characteristics can improve growth and survival.

Other abiotic factors important for lizards may also be influenced by habitat, for example humidity and substrate moisture can be affected by canopy cover, frequency of canopy gaps, sub-canopy vegetation density and leaf litter accumulation (von Arx *et al.*, 2012; Zhang & Pan, 2013). Appropriate humidity and substrate moisture levels are important in avoiding desiccation for some lizard species, particularly New Zealand endemic skinks (Neilson, 2002). Considered together, it is clear that the effects of habitat on microclimate play a critical role in lizard distribution.

3.1.2 – Habitat can influence lizard survival in the presence of predators

Another fundamental habitat feature that drives lizard distribution is the ability of a habitat to provide refuges from predators. It is well known that an important feature of lizard habitats is the presence of refuges that allow lizards to avoid or escape from predators (Smith & Ballinger, 2001; Block *et al.*, 2012). Studies have found that some species are capable of visually recognising those structures that provide increased protection (Schlesinger & Shine, 1994; Kerr *et al.*, 2003). In New Zealand, the only known mainland population of the endangered Whitaker's skink (*Oligosoma whitakeri*) has been found to be restricted to habitats that provide refuge from introduced predators (Towns & Elliott, 1996). Several studies have found that species richness of lizard assemblages tends to be higher in habitats with greater ground cover (Jellinek *et al.*, 2004; Garden *et al.*, 2007) and more structured near-ground strata (Hadden & Westbrooke, 1996; G. W. Brown, 2001; Garden *et al.*, 2007), which may be due to the increased predator protection these habitats provide. Both the vertical and horizontal components of habitat structure are vital for protection from predators. For example, in a vertically complex habitat, arboreal species can climb to escape terrestrial predators. A New Zealand study found that following the eradication of Pacific rats from Ohinau Island, Duvaucel's geckos used retreat sites that were on average 0.8m lower than prior to the eradication, suggesting that the geckos modified their use of the vertical habitat in the presence of predatory rats (Hoare *et al.*, 2007b). In contrast, rock crevices, woody debris or complex ground vegetation are important refuges for less arboreal species which may have limited ability to climb (East *et al.*, 1995). Horizontal vegetation is essential for both arboreal and smaller ground-dwelling species as it provides protection from avian predators (East *et al.*, 1995). In addition to providing refuge from predators, structurally complex habitats may hinder the movement of predators during predation attempts, allowing lizards to escape. Finally, many reptiles also require suitable retreats for long-term occupation during winter, when their activity is low or negligible (Gardiner *et al.*, 2013).

Due to the influence of habitat on lizard physiology, habitat parameters play an important role in determining lizard distributions. Additionally, it is likely that native lizards coexist with introduced mammals by avoiding habitats used by mammals, and by using refuges that mammals cannot penetrate, and as such habitat structure and composition may be important for coexistence. In order to determine how native lizards coexist with introduced rats, it is important to identify habitat features that may permit ground-dwelling lizards to persist in the presence of rats. This chapter aims to describe and compare the canopy and understorey structure and composition across six sites that support lizard populations of varying densities and species compositions. The parameters measured are listed in table 3.1, and were used to investigate how the habitats inhabited by lizards differ from habitats where lizards are absent.

Table 3.1 – Habitat parameters measured in this study and potential relationships with lizard presence.

Parameter	**Potential relationship with lizard presence**
Canopy tree density	Influences solar penetration, which in turn affects lizard thermoregulation.
Canopy cover	Influences solar penetration, which in turn affects lizard thermoregulation.
Canopy species diversity	More diverse forests are known to support more diverse faunas.
Important canopy species	Common canopy tree species have the greatest effect on leaf litter type and quality, which is particularly important for ground-dwelling skinks.
LSL structure	Highly structured environments may assist lizards in escaping from predators. Lizard species richness known to be associated with LSL structure.
LSL habitat type diversity	High diversity of habitat types provides different habitats for escaping and/or taking refuge from predators.
LSL cover	Influences solar penetration, which in turn affects lizard thermoregulation. Also provides protection from predators, particularly avian.

I test the following hypotheses that native lizard presence is associated with:

1) intermediate levels of canopy tree density and canopy cover. This is due to the interplay between habitat requirements for thermoregulation and protection from predators. These habitats would be open enough to allow patches of sunlight to reach the forest floor for lizards to bask in, but would provide enough cover for protection from high temperatures in summer and from predators.

2) high diversity of canopy tree species. Forests with high diversity of plant species are known to support more diverse faunas.

3) intermediate levels of lower shrub layer cover. Again, this is due to the interplay between habitat requirements for thermoregulation and protection from predators. A moderately dense lower shrub layer would be open enough to allow patches of sunlight to reach the forest floor for lizards to bask in, but would provide enough cover for protection from high temperatures in summer and from predators.

4) high diversity of lower shrub layer habitat types. A high diversity of habitat types provides a more structurally complex environment, which may assist lizards in escaping and taking refuge from predators.

5) larger amounts of leaf litter. I predict that leaf litter is a particularly important habitat type with regard to the ability of ground-dwelling native skinks to coexist with introduced predators. This is because these skinks have a limited ability to climb, and leaf litter may provide an important habitat that provides protection from predators.

3.2 – Methods

At each of the six sites (refer to section 2.2.2 for description of study sites), two 50 m by 50 m sampling grids were set up. Each grid was made up of three transects with three sampling points per transect, 25 m apart (18 sampling points per site, 108 sampling points in total). Habitat surveys were conducted at all sites between February 28 and April 5 2013. The following habitat information was collected at each sampling point.

3.2.1 – Canopy survey

Canopy surveys were conducted using the point-centred quarter method and tree density, basal area (cross-sectional area)and species importance values were calculated for each site as described in Mitchell (2010). Percentage canopy cover at each sampling point was estimated visually using a 50 x 50 cm quadrat, divided into 10 x 10 cm squares (plate 3.2).

Plate 3.2 - Canopy surveying. Left - measuring tree diameter as part of the point-centred quarter method. Centre and right - estimating canopy cover using a quadrat.

Canopy cover was tested for normality using Kolmogorov-Smirnov and Shapiro-Wilk tests, and differences between sites tested using Kruskal-Wallis (all in SPSS). Shannon's diversity index ($H' = -\sum_{i=1}^{R} p_i \ln p_i$ where p_i is the proportion of individuals belonging to the *i*th species) and Pielou's evenness index ($J' = \frac{H\prime}{H'max}$) were calculated for the species present at each site as in Onaindia *et al.* (2004).

3.2.2 – Lower shrub layer survey

The vertical structure of the lower shrub layer (hereafter LSL, includes vegetation up to 50cm height) was quantified using a cover board 50 cm high by 100 cm long, divided into 5 layers of 10 x 10cm squares. Observers moved 2 m from each sampling point in a random direction to avoid sampling disturbed vegetation caused by walking transects. The board was held upright at ground level by one observer and read by a second observer sitting or crouching so the board was at eye level (plate 3.3).

The number of squares touched by each habitat type (branches/fallen logs, ferns/moss, flax, herbs, leaf litter, seedlings, shrubs, trees and other) was recorded for each layer. The information used to classify the habitat types was obtained from published studies and supplemented by field observations prior to sampling. The ground layer (<10cm height as in Bee *et al.*, 1989; J. B. Wilson *et al.*, 1995) is likely to be the most important for ground-dwelling lizards such as skinks so the percentage cover for each habitat type was estimated for each of the 10 squares in this layer. Differences in LSL structure between sites was analysed using a Kruskal-Wallis test.

Plate 3.3 - Using the cover board to quantify lower shrub layer vegetation structure

Diversity and evenness indices were calculated for the habitat types at each site in the same manner as for the canopy species (section 3.2.1). Complexity scores were calculated for each sampling point using a modification of the method in Tasker and Bradstock (2006). For each layer, the number of squares on the cover board touched by each habitat type was added up. For layer A, the total number of squares contacted was multiplied by 5, for layer B the total was multiplied by 4 etc. The values for each of the 5 layers were summed to give an overall complexity score, and in this way greater importance was placed upon vegetation closer to the ground habitat of ground-dwelling skinks. Differences in average complexity scores between sites were analysed using a Kruskal-Wallis test.

The same quadrat that was used for estimating canopy cover was used to estimate horizontal (Christian *et al.*, 1983) LSL cover at each sampling point. The quadrat was held horizontally 0.5m above the ground by one observer and a second observer looked from directly above and visually estimated the percentage cover of each different habitat type (plate 3.4).

Plate 3.4 - Using the quadrat to estimate lower shrub layer cover

3.2.3 – Factor analysis

Patterns of covariation within the habitat variables were explored using principal component analysis with a varimax rotation using SPSS software (Version 21.0, IBM Corp). Rotations were carried out in order to obtain a set of uncorrelated factors with the simplest possible structure and rotated factor loadings are more interpretable than unrotated ones.

3.2.4 – Influence of habitat on lizard relative abundance

The factors identified in 3.2.3 were analysed with a Mann-Whitney test to determine whether there were difference in the habitats of sampling points with lizards and sampling points without lizards. Differences between sampling points with and without lizards were also examined using Bray-Curtis similarity (no transformation) and then non-metric Multi-Dimensional Scaling (MDS) (PRIMER 6, 2009). The relationship between the various habitat variables and native lizard relative abundance was investigated using Spearman correlations.

3.3 – Results

3.3.1 – Canopy characteristics and lizard abundance

The six sites used in this study showed variation in canopy cover and composition, although the important species across the six sites were similar with *Cyathea* (silver fern, or ponga)*, Kunzea* (kanuka, or white tea tree) and *Phyllocladus* (tanekaha, or celery pine) among the three highest importance values at all sites (see table 3.2).

Table 3.2 - Importance values of the canopy species at each site as determined by the point-centred quarter method. Importance values take into account relative density, cover and frequency. The three most important species at each site are shown in bold.

	Site						
SPECIES	**Eskdale**	**Kauri Glen**	**Kauri Park**	**Lady Phoenix**	**Silverdale Scenic**	**Torbay Heights**	**All Sites**
Cyathea	**51.28**	45.84	**45.84**	**94.36**	**97.68**	**67.98**	43.77
Kunzea	**57.94**	**70.23**	**119.07**	**41.38**	21.19	**48.18**	31.73
Phyllocladus	**45.50**	**48.54**	**83.06**		**36.41**	**36.13**	29.91
Myrsine	32.95	35.90	14.99	15.22	35.79	25.60	24.06
Pinus	36.26				**65.20**		20.34
Dicksonia			4.75				13.88
Agathis		**52.04**					13.74
Melicytus		7.20		**83.15**			11.40
Brachyglottis						4.65	9.24
Schefflera		3.67					8.95
Podocarpus					9.32		8.81
Coprosma	5.14			12.07	21.33	3.89	6.47
Knightia	28.27	3.95	5.17			5.37	6.42
Pseudopanax	12.90	3.26	11.04			11.61	6.22
Dacrycarpus		3.22				21.86	5.70
Pittosporum	4.31		4.29			5.64	5.57
Rhopalostylis						27.79	5.29
Leptospermum	3.59			15.60	5.27	11.56	5.26
Geniostoma				19.29	3.90	5.58	4.57
Weinmannia	11.18					16.16	4.33
Beilschmiedia		7.37	8.07				4.15
Cordyline		3.81		18.92			4.11
Acacia			3.72				3.85
Leucopogon					3.91		3.76
Hedycarya		3.58				3.79	3.30
Prumnopitys		3.31					3.17
Dysoxylum		3.28					2.78
Nestegis	3.59						2.63
Carpodetus		4.81				4.20	2.43
Olearia	3.57						2.42
Cyathodes	3.51						1.74

The sites with high to moderate abundance of native lizards tended to have higher canopy cover and greater species richness of canopy trees than the sites with low lizard abundance (table 3.3). The average percentage of canopy cover was highest at Eskdale Reserve (74 %), with Lady Phoenix and Silverdale Scenic Reserves having the lowest cover values (59% and 60%) (table 3.3).

Table 3.3 - Summary of canopy data across the six sites. Av % cover = average canopy cover of the 18 sampling points at each site. Values in parentheses indicate 95% confidence intervals.

Site	Native lizard abundance	Av % cover	Density (trees/ha)	Richness (no. spp)	Shannon-Wiener diversity	Pielou's evenness
Kauri Park	High	74 (65 - 83)	2,229	10	1.65	0.72
Eskdale	Moderate	72 (63 - 80)	2,289	14	2.22	0.84
Kauri Glen	Moderate	70 (62 - 79)	2,670	16	2.21	0.8
Torbay Heights	Moderate	59 (51 - 68)	2,712	16	2.4	0.86
Lady Phoenix	Low	60 (51 - 68)	2,595	8	1.77	0.85
Silverdale Scenic	Low	64 (56 - 72)	2,326	10	1.77	0.77

The canopy cover values were significantly right skewed (z-score = -5.04), indicating all sites had relatively high cover values, and both the Kolmogorov-Smirnov and Shapiro-Wilk tests were highly significant (KS = 0.185, df = 108, $p < 0.001$; SW = 0.915, df = 108, $p < 0.001$), indicating a non-normal distribution. The percentage of canopy cover was significantly different between sites (Kruskal-Wallis test, $p = 0.007$). Torbay Heights had the highest density of trees (2,712 trees/ha), while Eskdale Reserve and Kauri Park were the least dense sites but the difference between the most and least dense sites was just 483 trees/ha. Torbay Heights had the highest diversity of species of all six sites, both in terms of canopy species and LSL structure (tables 3.3 and 3.4), while Kauri Park and Silverdale Scenic Reserve had the lowest diversity in canopy species and LSL structure, respectively.

3.3.2 – Lower shrub layer characteristics and lizard abundance

Torbay Heights had the most highly structured LSL vegetation (72% of cover board squares contacted), as well as the highest average complexity (134.67) and diversity (1.97) scores, while Silverdale Scenic Reserve had the lowest values for these parameters (92.56 and 1.62) (table 3.4). Leaf litter was the most common habitat type encountered, making up an average of 32% of all vegetation at each site (figure 3.1). Shrubs (17%), branches/logs (15%), and herbs (14%) were also important components of the vegetation at all sites. Overall, approximately half of the vegetation in the LSL occurred in the lowest layer A (0 -10 cm off ground). A large proportion of this ground layer consistsed of debris (leaf litter and branches/logs) rather than living plants (table 3.4 and figure 3.2).

Table 3.4 - Summary of lower shrub layer vegetation structure. % contacts = percentage of squares on the cover board contacted by all types of vegetation, including leaf litter. Note that the % contact values allow for superimposition of several habitat types onto one square and therefore values greater than 100% are possible. Layer A = 0-10cm from ground level.

Site	Lizard abundance	% contacts All layers	% contacts Layer A	Average complexity score	Shannon-Wiener diversity	Pielou's evenness
Kauri Park	High	59.56	35.78	122.61	1.66	0.76
Eskdale	Moderate	61.22	30.44	114.94	1.87	0.85
Kauri Glen	Moderate	59.56	32.44	118.17	1.83	0.83
Torbay Heights	Moderate	72.22	32.22	134.67	1.97	0.9
Lady Phoenix	Low	47.89	29.33	99.17	1.7	0.77
Silverdale Scenic	Low	45.56	28.11	92.56	1.62	0.74

The sites with moderate to high abundance of native lizards (Kauri Park, Eskdale Reserve, Kauri Glen and Torbay Heights) generally had higher LSL structure (% contacts), complexity and diversity (table 3.4). Conversely, the sites with the lowest abundance of native lizards tended have much lower values for these habitat parameters.

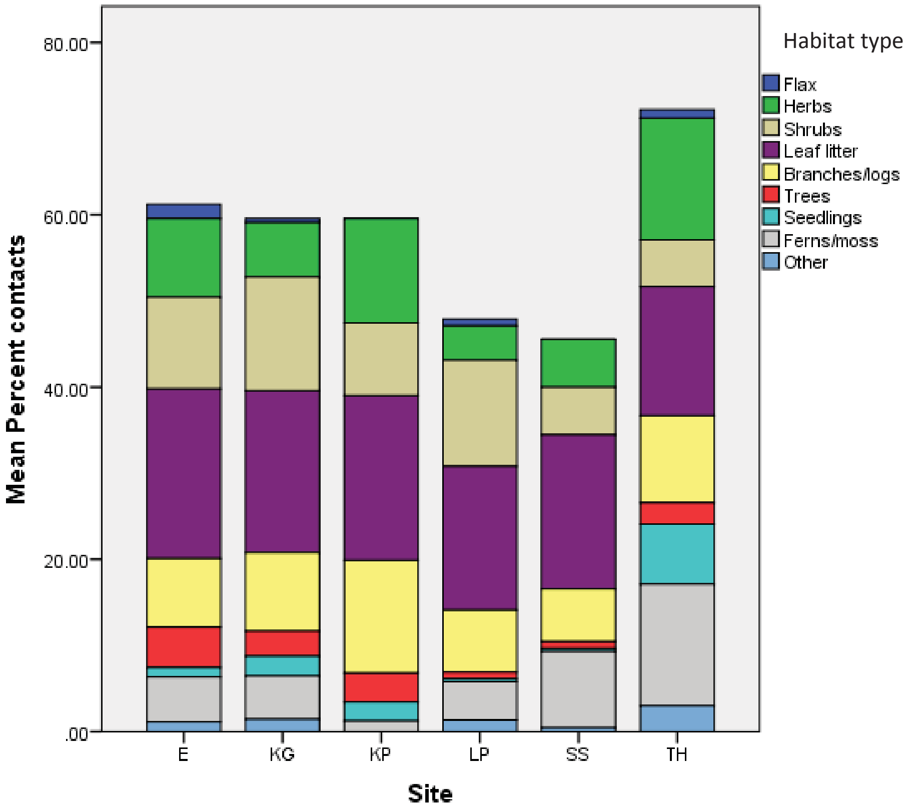

Figure 3.1 - Vegetation structure of the lower shrub layer. E = Eskdale, KG = Kauri Glen, KP = Kauri Park, LP = Lady Phoenix, SS = Silverdale Scenic, TH = Torbay Heights.

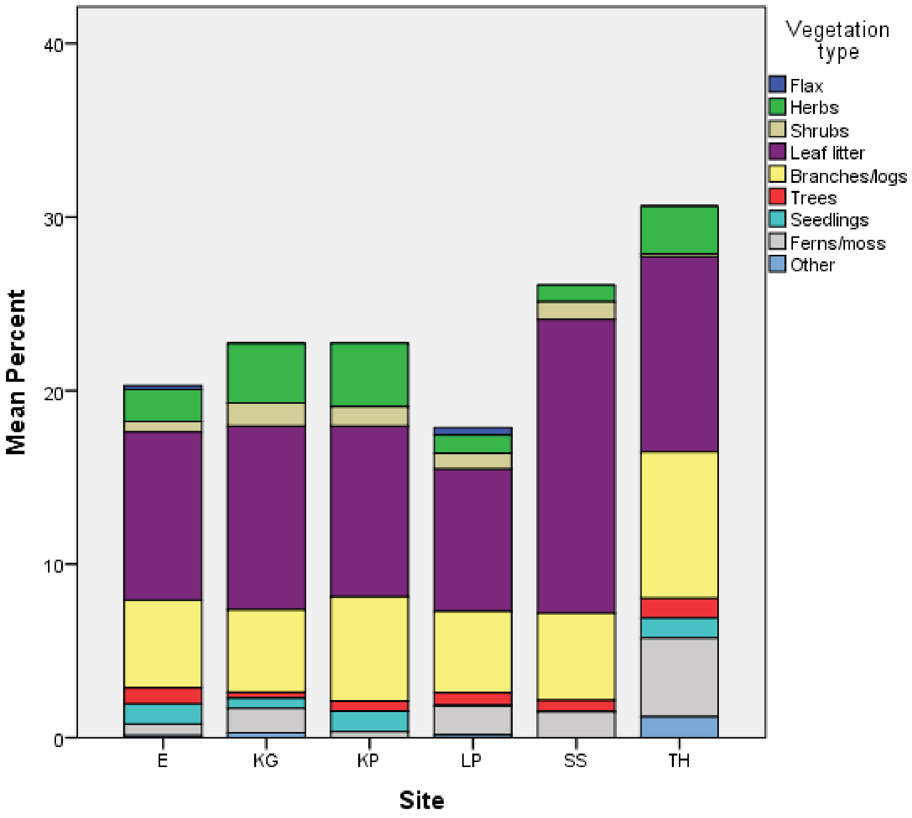

Figure 3.2- Vegetation composition of the ground layer (layer A, lowest 10cm). E = Eskdale, KG = Kauri Glen, KP = Kauri Park, LP = Lady Phoenix, SS = Silverdale Scenic, TH = Torbay Heights.

In terms of LSL cover, leaf litter was again the most frequent habitat type encountered at all sites (table 3.5), with branches/logs and herbs also being important vegetative features (figure 3.1). Torbay Heights had the highest overall vegetation cover, while the lowest values for LSL cover were seen at Kauri Park and Kauri Glen. The results of the Kruskal-Wallis test for the percentage of squares touched (all layers) and the average complexity scores were highly significant (p = 0.000 and p =0.001 respectively) indicating that both variables were significantly different between sites. Most sites showed similar patterns in LSL composition except for Torbay Heights, where ferns/moss and herbs were almost as dominant as leaf litter, and the frequency of seedlings was also very high compared with other sites (figure 3.1).

Three of the sites with moderate to high native lizard abundance (Eskdale Reserve, Kauri Glen and Kauri Park) tended to have higher leaf litter cover and lower vegetation cover than the sites with low lizard abundance (table 3.5).

Table 3.5 - Summary of lower shrub layer vegetation cover. Debris = leaf litter and branches/logs; Vegetation = cover of all other habitat types excluding leaf litter and branches/logs.

Site	Lizard abundance	Av % debris cover	Av % vegetation cover	Av % bare earth
Kauri Park	High	74.89	17.5	7.61
Eskdale	Moderate	77	20.11	2.89
Kauri Glen	Moderate	76.39	19.17	4.44
Torbay Heights	Moderate	60.78	34.78	4.44
Lady Phoenix	Low	66.78	28.89	4.33
Silverdale Scenic	Low	69.5	28.28	2.22

3.3.3 – Factor analysis

The final factor analysis resulted in two components being selected – one including the LSL variables and canopy cover, and the other relating to canopy tree basal area and density (table 3.5). Together these components account for 59.4% of the variance in the habitat variables. The initial factor analysis also included ground layer vegetation and debris, which were removed as they correlated extremely highly with other variables and as such the ground layer variables will not be analysed further.

Table 3.6 – Rotated component matrix for the factor analysis of habitat variables. Component scores below 0.4 are not shown for clarity. LSL = lower shrub layer; debris = leaf litter and branches/logs; vegetation = ferns/moss, flax, herbs, seedlings, shrubs, trees and other; % contacts = percentage of squares on cover board contacted; BA = basal area of trunk; distance = distance to nearest tree.

	Eigenvalue	
	Component 1	Component 2
LSL % vegetation cover	-.908	
LSL% debris cover	.904	
LSL % contacts by vegetation	-.718	
LSL% contacts by debris	.500	
Canopy cover (%)	.453	
Canopy tree average distance (cm)		.803
Canopy tree average BA (cm²)		.774

3.3.4 – Influence of habitat on lizard relative abundance

Mann-Whitney tests did not reveal any significant differences between sampling points with and without lizards for factor 1 (p = 0.703) or factor 2 (p = 0.057). This was corroborated by the error bar graphs, which show large overlaps for both factors (figure 3.3).

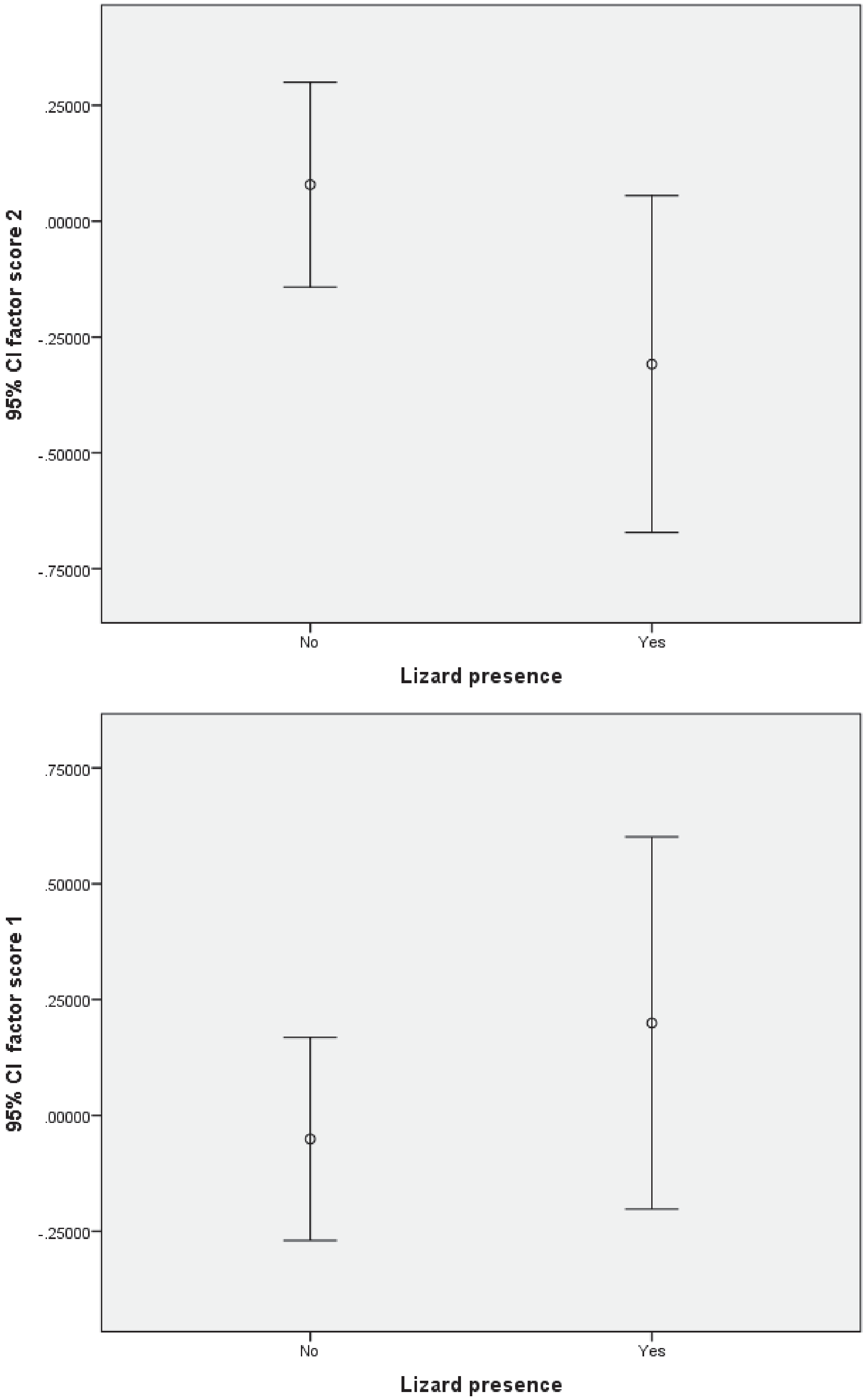

Figure 3.3 - Error bar graphs of the factor scores for the habitat variables. Factor 1 includes LSL vegetation cover, LSL debris cover, LSL vegetation structure, LSL debris structure and canopy cover.

MDS did not reveal any differences in the habitats of sampling points with lizards and those without lizards (figure 3.4).

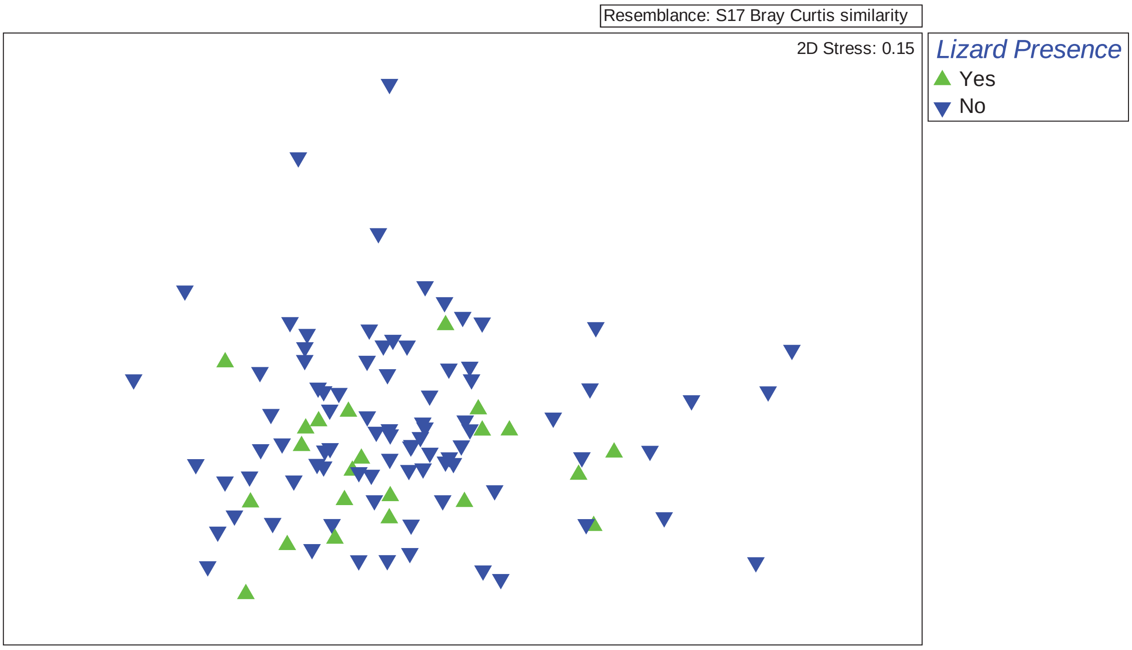

Figure 3.4 - MDS plot showing habitat variables of sampling points with and without lizards.

However, looking at the site level did reveal some significant relationships (table 3.6). Spearman correlations between native lizard relative abundance and the seven habitat variables identified by factor analysis (section 3.3.3) revealed a significant positive relationship with LSL debris structure.

Table 3.7 - Spearman correlations between native lizard relative abundance and various habitat variables. LSL = lower shrub layer; debris = leaf litter and branches/logs; vegetation = ferns/moss, flax, herbs, seedlings, shrubs, trees and other; % contacts = percentage of squares on cover board contacted; BA = basal area of trunk; distance = distance to nearest tree. Bonferroni corrwction applied, therefore sig = 0.007, significant results denoted with *.

Habitat variable	r	*p*	
Canopy cover (%)	0.812	0.050	
LSL% debris cover	0.638	0.173	
LSL % vegetation cover	-0.812	0.050	
LSL % contacts by vegetation	0.406	0.425	
LSL% contacts by debris	0.986	0.000	*
Canopy tree average distance (cm)	0.464	0.354	
Canopy tree average BA (cm²)	0.232	0.658	

3.4 – Discussion

3.4.1 – Canopy characteristics and lizard abundance

The results of this study did not support my hypothesis that intermediate levels of canopy tree density and canopy cover would be associated with native lizard presence. In fact, the sites with high to moderate native lizard presence had higher canopy cover than the sites with low lizard abundance, while density didn't show any discernible pattern with relation to lizard abundance. One potential explanation for the association of native lizards with relatively high canopy cover is that the effect of canopy cover on thermoregulation is not as important in determining lizard presence as the ability of the habitat to provide protection from predators, particularly in predator-dense urban forest remnants.

The differences in canopy cover and canopy tree density between sites may be explained by differences in the age of the forest fragments and the presence of pine trees (*Pinus spp.*). Lady Phoenix Reserve for example, is young and shrubby and has recently been replanted in some areas. In some parts the forest lacks a canopy entirely and resembles a shrubland with the occasional emergent tree. Silverdale Scenic Reserve is also fairly young but an important aspect here is the prevalence of *Pinus* trees (which had the second highest importance value at this site, Appendix II), that has resulted in many canopy gaps due to falling trees and branches (pers. obs.). In comparison, the forests at Eskdale Reserve, Kauri Glen and Kauri Park are older and have well-developed canopies with comparatively fewer gaps and therefore higher canopy cover values. These observations are in line with other studies which found canopy cover to increase with age (Halpern & Spies, 1995). The age of the stands may also explain the varied tree density values across the sites. The more mature sites, Eskdale Reserve and Kauri Park, have the lowest densities of trees. These sites have well developed canopies which mean less light reaching to lower layers to allow growth of seedlings, which results in a lower frequency of larger trees, as opposed to a higher number of small trees as at Lady Phoenix and Silverdale Scenic Reserves (Diaci *et al.*, 2012; Donoso *et al.*, 2013).

It is interesting that in this study native lizards appear to be associated with older sites with greater canopy cover. Sites with few canopy gaps and a full canopy have different solar regimes compared with less mature forest patches (Holeksa, 2003) and will experience lower temperatures and less solar penetration. As diurnal heliotherms, native lizard species might be expected to be more abundant in more open habitats with greater access to sunlight for basking and less abundant in areas with closed canopies, however the results of this study suggest the opposite. One potential explanation for this could be competitive exclusion by the introduced Australian rainbow skink. Among the six study sites, those with more open patches such as Silverdale Scenic Reserve and Torbay Heights tended to be warmer and drier, especially in summer, and had comparatively greater numbers of rainbow skinks and fewer native species than patches with more closed canopies such as Kauri Park and Eskdale Reserve (section 2.3.1). The rainbow skink's robustness to disturbance, high reproductive rate and generalist diet mean that they may be able to outcompete native species, perhaps to the point where natives are excluded from these more open, sunny areas. It may be that native skinks do not prefer greater canopy cover per se, but are more abundant in these areas where they are not competing with the rainbow skink. A second potential explanation is the sensitivity of some native species to evaporative water loss (Neilson, 2002), which would be greater in warmer, more exposed patches and consequently may discourage native species. The rainbow skink, in contrast, may be better able to tolerate these conditions as it is adapted to the generally warmer and more arid conditions of its native country, Australia. A third possible explanation may be that lizards are not associated with higher canopy cover per se, but perhaps another habitat feature that is related to canopy cover, such as the cover of leaf litter. More mature sites with greater canopy cover may have greater cover of leaf litter as a direct result of the presence of numerous, large trees which drop leaves and other debris. In predator-rich environments with little cover of low-growing vegetation, such as forests, leaf litter may be an important refuge for native lizards that allows them to escape predation attempts and thereby coexist with introduced mammals.

The second hypotheses regarding the association of native lizards with a high diversity of canopy tree species was also not supported. The three sites with moderate native lizard abundance had the highest diversity of canopy tree species, and the sites with low native lizard abundance had lower diversity of canopy species but interestingly the site with the highest native lizard abundance, Kauri Park, had the lowest canopy tree diversity. If Kauri Park were to be excluded, there would be a general pattern of increasing lizard abundance with canopy tree species diversity.

The six sites were very similar in terms of the important canopy tree species present, with a few notable differences. *Agathis australis*, or Kauri, has a high importance value at Kauri Glen, possibly due to this species tending to have very a large trunk diameter which contributes to the cover value used to calculate the importance score. Although *Agathis* tends to have relatively low frequencies, where it is present it tends to be a conspicuous element of the canopy and also the leaf litter, which may be more relevant for ground-dwelling skinks. Interestingly, three of the sites with moderate to high abundance of native lizards were kauri forests (Kauri Park, Kauri Glen and Eskdale Reserve). *Pinus* also tends to be important when it is present – it has the second highest importance value at Silverdale Scenic Reserve and fourth highest at Eskdale Reserve – and probably for a similar reason to *Agathis* as *Pinus* also tends to have a very large trunk diameter. The presence of *Pinus* in particular may have an impact on the surrounding vegetation due to the increased soil acidity associated with pine needle fall, which also affects invertebrate populations and potentially lizards too (Cakir & Makineci, 2013; Hizal *et al.*, 2013). Also notable is the comparatively high importance value of *Melicytus* (mahoe or whiteywood) at Lady Phoenix Reserve compared to the other sites. *Melicytus* is frequent and dense here – two of the components used in calculating the importance value. This is likely to be a reflection of the age of this very young forest patch which has low canopy cover and some patches of shrubland that lack a canopy altogether. The dominance of Melicytus may be another reason that native lizards are not abundant here, as the quality of leaf litter provided by *Melicytus* is poor terms of its ability to provide lizard habitat. The main body of fallen *Melicytus* leaves tend to quickly decompose, leaving only the 'skeleton' of the veins, which provides very little cover for lizards (pers. obs.).The presence of the shrubland at Lady Phoenix Reserve highlights a potential limitation to using the point-centred quarter method for this site. Only trees with a trunk diameter of 4cm or higher are measured, and so many young trees or thin-trunked shrubs would be excluded, although they may still be important especially in the absence of a true canopy.

The high vegetation diversity (both for canopy and LSL) at Torbay Heights is mainly due to the diverse range of habitats and forest types present here. Sampling grid one includes patches of relatively young bush dominated by common species like *Phyllocladus, Kunzea* and *Cyathea,* then changes to open scrub with pampas grass (*Cortaderia selloana*), tangle fern (*Gleichenia dicarpa*) and only a few trees, including exotic pine (*Pinus spp*). Grid two contains a swampy area dominated by kiekie (*Freycinetia banksii*), nikau (*Rhopalostylis sapida*), kahikatea (*Dacrycarpus dacrydioides*) and supplejack (*Rhipogonum scandens*), which is a stark contrast to the diverse forest that makes up the remainder of the grid and includes species such as kamahi (*Weinmannia racemosa*), rewarewa (*Knightia excelsa*), pigeonwood (*Hedycarya arborea*) and rangiora (*Brachglottis repanda*). This very high diversity of habitat types may have partially obscured some relationships between habitat and native lizard abundance. For example, native lizards were only found on grid two, which had a greater canopy cover than grid one and was structurally more similar to the sites with higher relative abundance of native lizards (pers. obs.). Grid one, on the other hand, lacked a canopy in many places and was quite different to any of the other habitats both structurally and floristically. Looking at Torbay heights as a whole, the habitat at grid one probably reduced the overall canopy values at the site although native lizards were never found on this grid and it had the highest numbers of rainbow skinks.

In contrast to Torbay Heights, Kauri Park has the lowest canopy species diversity of all the sites and the highest abundance of native lizards. Here, the three most dominant species (*Phyllocladus*, *Kunzea* and *Cyathea*) make up over 80% of canopy trees, while at Torbay Heights the three most dominant species (*Cyahea, Phyllocladus* and *Kunzea*) make up less than 50% of the canopy. The Shannon diversity index used takes species evenness into account, so the dominance of a few species partly explains the low diversity at Kauri Park. This site also has lower species richness - 10 species compared to 16 species at Torbay Heights – which further contributes to its low diversity. It may be that floristic diversity of the canopy is less important to ground-dwelling native lizard species than the structural attributes such as canopy cover and density of canopy trees, which in turn have an impact on other habitat attributes, such as solar penetration, temperature and humidity.

3.4.2 – Lower shrub layer characteristics and lizard abundance

The hypothesis that native lizard abundance would be associated with intermediate levels of lower shrub layer cover was not supported by my results, however the results did support the fourth hypothesis that native lizard abundance would be associated with a high diversity of lower shrub layer habitat types, and therefore increased structural complexity. Generally, the sites with high to moderate relative abundance of native lizards had relatively high values for LSL structure (% contacts), complexity and diversity, while the sites with low relative lizard abundance had very low values for these parameters. This may suggest that the ability of the habitat to provide protection from predators (and LSL vegetation is particularly relevant for ground-dwelling predators such as rats) is more crucial for lizard survival in urban forest fragments than the thermoregulatory features of the habitat. This is also consistent with the results of other studies which have found that structurally diverse habitats may be advantageous for lizard populations by providing opportunities for escaping and taking refuge from predators (Martin & Lopez, 1995), and allowing for seasonal variation in thermoregulation and refuge requirements (Christian *et al.*, 1983; Gardiner *et al.*, 2013) and can support more abundant and varied lizard communities (East *et al.*, 1995).

I also hypothesised that the native lizards would be associated with habitats with larger amounts of leaf litter, which was supported by my results. The habitat surveys revealed that half of the vegetation in the LSL is in the lowest 10cm, and a sizeable proportion of this is leaf litter and debris. This may partly explain why native lizards were more abundant at sites with greater leaf litter cover (Eskdale Reserve, Kauri Glen and Kauri Park), as the structurally complex leaf litter and branches/logs at ground level would provide refuge from both avian and terrestrial predators. Studies of other ground-dwelling lizard species have shown that some species do prefer to use leaf litter as a means of predator avoidance, and suggest that some species actively select leaf litter with characteristics that maximise camouflage from predators (Valentine *et al.*, 2007; Wall & Shine, 2013). The use of leaf litter as a refuge for lizards may be particularly important in mature forests with reduced vegetation for refuges in the understorey due to light limitation.

3.4.3 – Factor analysis

The components extracted by the factor analysis showed some interesting features. Component 2 for example, contains two variables: basal area and average distance from the sampling point to the nearest canopy tree. This component appears to be related to forest age, as larger basal areas tend to be associated with older trees and the average distance between trees tends to increase as forests age, older trees die and canopy species overshade seedlings and reduce growth in the understorey. Component 1 reveals some interesting relationships between the five habitat variables it contains. Canopy cover is related to the amount of debris found below, because a forest with greater canopy cover will also contribute more leaf litter and fallen branches to the LSL. It also makes sense that the opposite relationship would be true for vegetation cover and structure, as canopy cover increases we might expect vegetation in the LSL to be limited by reduced light levels.

3.4.4 – Influence of habitat on lizard relative abundance

Although some interesting patterns were observed between native lizard abundance and individual habitat parameters, comparison of the habitats at sampling points with lizards and without lizards did not give any significant results. The main reason for this is probably the very low capture efficiency for lizards in this study. As a result most sampling points had no lizard captures, although the habitats at these points may have been used by lizards that were simply not detected. As such, any relationships between habitat parameters and lizard abundance would have been obscured.

However, looking on a larger scale at the sites rather than the sampling points did reveal some interesting relationships. As previously discussed, while the positive relationship between canopy cover and native lizard relative abundance is not what I expected, there are several possible explanations as to why this relationship occurs (section 3.4.1). Greater canopy cover may result in greater cover and structure (depth) of leaf litter also, which could be an important refuge for native lizards that allows them to coexist with introduced mammals. This would also explain the positive relationship of native lizard relative abundance with LSL debris structure, which includes leaf litter and branches/logs.

Chapter 4

Invertebrate communities and the abundance of ground-dwelling lizards in urban forest fragments

Plate 4.1 - One of the large spiders caught during invertebrate sampling.

4.1 – Introduction

4.1.1 – The importance of invertebrates as a food source for native lizards

Invertebrates are known to make up a large proportion of the diets of native lizards, although the exact proportion can vary considerably between species and across seasons. The proportion of invertebrate food items in the diets of grand and Otago skinks (*Oligosoma grande* and *O. otagense*), for example, can vary from 80% and 82% respectively in November, to 47% and 38% respectively in May (Tocher, 2003) (Gill, 1976; Porter, 1987; Patterson, 1992; Spencer *et al.*, 1998; Tocher, 2003). The invertebrate component of native lizard diets includes a wide variety of groups including Diptera (true flies), Coleoptera (beetles), Araneae (spiders), Acari (mites and ticks), Opiliones (harvestmen), Orthoptera (grasshoppers and weta), Diplopoda (millipedes), Hymenoptera (ants) and Isopoda (woodlice) (Gill, 1976; Patterson, 1992; Spencer *et al.*, 1998; Tocher, 2003).

There has only been one previous study on the diet of copper and ornate skinks in the Auckland area (Porter, 1987). This study found that Acari, Araneae and Coleoptera are the most important invertebrate groups in the diets of both species, with shelled mollusca also being important in the diet of ornate skinks. It was also found that ornate skinks took larger prey items than copper skinks did due to its larger body size (Porter, 1987), which is supported by the findings of other studies on prey size selection in native skinks (Montoya & Burns, 2007).

However, like most published studies regarding the diets of native skinks (Patterson, 1992; Spencer *et al.*, 1998; Tocher, 2003), it was undertaken in grassland habitats and therefore may not be representative of the diets of lizards in forest habitats.

4.1.2 - Food availability affects lizard populations

Food availability has important impacts on lizard growth, fecundity, distribution, foraging behaviour and predation risk (Dunham, 1978; Guyer, 1988; Pastro *et al.*, 2013). Spatial and temporal distributions are two aspects of lizard ecology that are known to be influenced by food availability. In a study of a common Mediterranean lizard, *Psammodromus algirus,* it was found that their distribution was partly explained by food availability in the form of arthropod abundance, which was itself influenced by habitat factors (see 3.1.3) (Diaz & Carrascal, 1991). Lizard captures in the Amazon rainforest have also been found to be strongly associated with the presence of food in the form of termite nests (Garda *et al.*, 2013), again showing the impacts of food availability on the spatial distributions of lizards. The role of food availability in temporal distributions of lizards is less well studied, however. Some native gecko species are known to use flowering trees for food in the form of nectar (Eifler, 1995), and as the flowering of the tree species involved is highly seasonal, it is likely that the temporary availability of this energy-rich food source has some impact on lizard distribution on a temporal scale.

One of the most important influences of food availability on lizard populations is the trade-off between foraging and predation risk, and the subsequent impacts on energy gain and expenditure (Huey & Pianka, 1981). A habitat with a high density of food items allows lizards to reduce their total foraging effort, minimizing activity that may expose them to predators. A habitat with larger food items can have a similar effect by reducing the number of foraging events required for lizards to meet their energy needs and thereby also reducing their predation risk (Pough & Andrews, 1985). A habitat with a high density of food items, or with larger food items, may therefore facilitate coexistence of lizards and predators better than a habitat with lower density or smaller food items by reducing the exposure of the lizards to predators. Food availability and distribution have also been shown to affect the foraging behaviour of native lizards. For example, experimental manipulation of prey distribution was found to alter the duration and frequency of foraging movements of native grand skinks (Eifler & Eifler, 1999), which in turn is likely to affect energy budgets and predation risk.

However, a high density of invertebrates may also have negative impacts on lizard populations by supporting higher densities of rats, which then exert greater predation pressure on the lizards.

4.1.3 – Factors affecting invertebrate availability

In order to understand the effect of food availability on lizard populations, it is important to look at the factors that shape food availability. In the case of invertebrates, their availability may be affected by human disturbance, competition with other species, and habitat factors such as vegetation cover and microclimate (Evans *et al.*, 2003;

Buckingham *et al.*, 2006; Tulp & Schekkerman, 2008; Norbury *et al.*, 2009). Positive relationships have been found between vegetation ground cover and invertebrate captures in a dryland ecosystem in New Zealand (Norbury *et al.*, 2009). In this study, few beetles were caught in pitfall traps when cover was less than 80%, no millipedes were caught where cover was below 70%, and few spiders were caught where vegetation cover was less than 50%.

Human activities can also have a considerable effect on invertebrate communities, and subsequently food availability for insectivorous lizards. One study found that the edge habitat provided by the presence of roads within forests had an impact upon the invertebrate community. It was found that road-side invertebrate communities had higher numbers of introduced species than forest interior communities (Delgado *et al.*, 2013), which may in turn have an impact on the availability of food for lizards if they prefer native species over introduced species.

Food availability for lizards may also be affected by competition with other species, both native and exotic. Rodents, for example, are likely to compete with native lizards for invertebrate prey, with studies showing that invertebrates are often the major component of rat and mouse diets (being present in over 90% of rat stomachs and 80% of mouse stomachs) (Taylor & Thomas, 1993; Miller & Miller, 1995). The presence of rodents on islands off the coast of New Zealand is responsible for serious declines and extirpations of resident invertebrate populations, which may in turn reduce food availability for native lizards (Bremner *et al.*, 1984; Newman, 1994; Towns, 2009; St Clair, 2011; Ruscoe *et al.*, 2013). It was reported that 15 species of invertebrates from many groups including spiders, beetles, weta, earwigs and snails were extirpated or declined due to rats, and in many cases the populations were observed to reappear or recover following the eradication of rats (Towns, 2009). Rodents are not the only introduced mammals that are likely to compete with native lizards for invertebrate prey. It has also been shown that hedgehogs prey upon native invertebrates such as weta, and that their impact on the prey population is proportional to the density of the hedgehog population (Jones *et al.*, 2013).

In addition to the introduced small mammals, introduced lizards may also compete with native species for invertebrate prey. In New Zealand, the introduced rainbow skink has been suggested as a potential competitor for native skinks such as the copper skink, based on the overlap in distribution, microhabitat use, foraging strategies, activity periods, prey type and size (Peace, 2004). Rainbow skinks have also been introduced to the islands of Hawaii, where they are quickly expanding in range and appear to be replacing the native moth skink (*Lygosoma noctua noctua*) (Hunsaker & Breese, 1967). Despite the rainbow skink being suggested as a potential competitor by numerous governmental and herpetological organizations (including the New Zealand Herpetological Society, the Ministry for Primary Industries and the Auckland Council), there have only been two studies directly investigating this theory – one found that introduced rainbow skinks could successfully compete with native copper skinks for food (West, 1979), while the other found no evidence of competition (Peace, 2004). However, until a mechanism that allows coexistence is identified, competition between the native and introduced species cannot be excluded.

Given that invertebrates make up a significant proportion of the diet of native lizards, and that food availability is known to influence the distribution of other reptile species, it is a logical extension that invertebrate availability may have an impact on the abundance and distribution of native lizards in urban forest fragments. In predator-rich urban environments such as Auckland, invertebrate availability may also influence the ability of native lizard species like the copper and ornate skinks to coexist with introduced mammals through the trade-off between foraging and predation risk. As well as being predators of native lizards, introduced mammals such as rats, mice and hedgehogs, are also competitors for invertebrate prey. Urban forest fragments in Auckland are also home to the introduced rainbow skink, which is another potential competitor for native lizards that may impact on their ability to coexist with introduced mammals. In this part of my study I aim to compare invertebrate abundance and diversity between sites and investigate the effects on lizard distribution. I will also investigate the potential effects of competitors on invertebrate availability by comparing invertebrate abundance between sites with varying competitor densities. Finally, I will investigate the survey methods used and potential intra-seasonal changes in invertebrate abundance by comparing invertebrate abundance over time and between the different baits used in the traps.

4.2 – Methods

The six study sites within urban forest fragments and survey grid layout are as described in section 2.2.2. The modified pitfall trap design, baits used, and trapping regime are as described in sections 2.2.3 and 2.2.4 (see also figure 2.8). Invertebrates were live captured, identified and counted in the field. Although it is standard practice to use so kind of solution to kill/preserve invertebrates caught in traps, it was not used here as invertebrate trapping had to coincide with lizard trapping and any solution used would also kill lizards. The limitations of this are discussed further in section 4.4. Captured invertebrates were assigned to one of the six size classes listed in table 4.1. Specimens that were less than 1 mm in size could not be reliably identified and were therefore excluded from analysis. Any invertebrates that could not be identified in the field were taken back to the lab for identification. Invertebrates were generally identified to order or superorder, although some were identified to higher taxonomic levels where identification to lower classification levels was problematic or not relevant for lizard diets. The taxonomic groups used to classify invertebrates in this study are listed below in table 4.2. After being identified and counted, invertebrates were released 2 m from the traps to avoid recaptures on subsequent nights.

Table 4.1 - Invertebrate body size classes. Note that invertebrates with body sizes < 1 mm were not counted.

Size class	Invertebrate body size (mm)
1	1 – 2
2	3 – 10
3	11 – 20
4	21 – 30
5	31 – 40
6	≥ 40

Table 4.2 - Invertebrate groups identified in this study.

Taxon	Taxonomic level	Common name
Acari	Subclass	Mites, ticks
Amphipoda	Order	Landhoppers
Araneae	Order	Spiders
Archaeognatha	Order	Jumping bristletails
Chilopoda	Class	Centipedes
Coleoptera	Order	Beetles, weevils
Collembola	Order	Springtails
Dermaptera	Order	Earwigs
Dictyoptera	Superorder	Roaches, mantids
Diplopoda	Class	Millipedes
Diptera	Order	Flies, mosquitoes, gnats
Gastropoda	Class	Terrestrial snails, slugs
Hemiptera	Order	Tree bugs, cicadas, aphids, spittlebugs
Hymenoptera	Order	Ants, bees, wasps
Isopoda	Order	Slaters
Lepidoptera	Order	Butterflies, moths
Nematoda	Phylum	Round worms
Oligochaeta	Subclass	Earth worms
Opiliones	Order	Harvestmen
Orthoptera	Order	Grasshoppers, locusts, katydids, crickets
Pseudoscorpionida	Order	False scorpions
Symphyla	Class	-
Thysanura	Order	Silverfish, bristletails
Turbellaria	Class	Terrestrial flatworms

The invertebrate communities at each site were compared in terms of invertebrate abundance, richness and diversity. The Shannon-Weiner diversity index ($H' = -\sum_{i=1}^{R} p_i \ln p_i$, where p_i is the proportion of individuals belonging to the *i*th species) and Berger-Parker index (dominance index, maximum p_i) were calculated for each site. In addition to investigating total invertebrate abundance, the abundance of likely prey groups was compared between sites. Likely prey groups were predicted to be Acari, Araneae and Coleoptera (based on dietary analysis of copper and ornate skinks by Porter (1987)) that were < 10 mm in size (based on the prey sizes of two congeneric native skink species reported by Gill (1976) that have similar body sizes to the two species in this study).

Differences in invertebrate assemblages were examined using Bray-Curtis similarity (with square root transformation) and then MDS (non-metric Multi-Dimensional Scaling) analysis (PRIMER 6, 2009). One-way ANOSIM (analysis of similarity) was then run to assess assemblage differences (PRIMER 6, 2009). These tests were used to compare samples between sites, sampling sessions and the baits used in the pitfall traps. Chi-squared tests of homogeneity were carried out in Microsoft Excel to examine differences between sampling sessions at each site, and differences between baits. For these analyses, only taxa that were common at most sites and were important food taxa were considered. To counteract the multiple comparisons, a Bonferroni correction was applied in each case.

Potential effects of competitor abundance on invertebrate abundance at each site were investigated using a Spearman's rank correlation in SPSS. Spearman's rank correlation was also used to investigate the relationship between native lizard relative abundance and invertebrate abundance. Invertebrate abundances from the third invertebrate sampling session only were used, as mammal trapping was undertaken immediately following this sampling session.

4.3 – Results

4.3.1 – Comparison of invertebrate abundance and diversity between sites

A total of 42,076 invertebrate specimens from 24 different taxa were captured and identified over 1,944 trap nights. Most sites had fairly similar invertebrate richness (20 – 22 taxa present), except Silverdale Scenic, which had only 17 taxa (table 4.3). The site with the highest lizard abundance, Kauri Park, had moderate invertebrate abundance, richness and diversity. The number of invertebrates captured at each site varied widely, from 4,073 individuals at Silverdale Scenic Reserve to 13,454 at Lady Phoenix Reserve (table 4.3). The very high numbers of invertebrates caught at Lady Phoenix Reserve and Torbay Heights are due to unusually high numbers of Hymenoptera, ants in particular (table 4.4) (only seven of the 30,467 Hymenoptera captured in this study were not ants). This is also reflected in the high dominance values for both of these sites.

Table 4.3 - Summary of invertebrate abundance, richness and diversity across the six study sites. See table 4.2 for a full list of taxa identified in this study. Lizard abundance estimates from section 2.3.1.

Site	Native lizard abundance	Total number of individuals	Number of taxa present	Shannon-Wiener diversity index	Berger-Parker index
Kauri Park	High	4,727	21	1.24	0.68
Eskdale	Moderate	5,564	21	1.44	0.58
Kauri Glen	Moderate	4,798	20	1.55	0.57
Torbay Heights	Moderate	9,460	22	0.99	0.78
Lady Phoenix	Low	13,454	22	0.76	0.84
Silverdale Scenic	Low	4,073	17	1.2	0.65
TOTAL	-	42,076	24	-	-

The extremely high incidence of Hymenoptera also explains why the most species-rich sites (Lady Phoenix Reserve and Torbay Heights) were the least diverse according to the Shannon-Wiener index (0.76 and 0.99, respectively), which takes into account both richness and evenness (table 4.3). This is corroborated by the Berger-Parker index, which shows that the invertebrate communities at Lady Phoenix Reserve and Torbay Heights are very uneven, with Hymenoptera being the most abundant species at these sites and making up 84% and 78% of the total catch, respectively (tables 4.3 and 4.4). Conversely, the most diverse sites, Kauri Glen and Eskdale Reserve had the lowest proportions of Hymenoptera, at 58% and 57% respectively, which resulted in these sites having high diversity (1.55 and 1.44, respectively) despite having slightly lower species richness.

Hymenoptera and Coleoptera were the two most abundant taxa overall, and were among the five most abundant taxa at all six sites (table 4.4). Collembola, Araneae and Amphipoda were also common, being among the five most abundant taxa at five of the six sites. Of the 24 taxa identified, 17 taxa each comprised less than 1 % of the total invertebrate captures.

Table 4.4 – The percentage of total catch at each site represented by each taxon, from most abundant over all six sites to least abundant. The five most common taxa at each site (by percentage) are indicated in bold.

Taxon	**Eskdale**	**Kauri Glen**	**Kauri Park**	**Lady Phoenix**	**Silverdale Scenic**	**Torbay Heights**	**All Sites**
Hymenoptera	**57.66**	**57.21**	**67.65**	**84.02**	**65.16**	**77.78**	68.25
Coleoptera	**10.21**	**10.53**	**11.36**	**5.83**	**18.78**	**6.25**	10.49
Collembola	**17.92**	**4.98**	**3.72**	**3.38**	**3.76**	1.47	5.87
Amphipoda	2.23	**11**	**8.29**	0.83	**3.76**	**4.82**	5.15
Araneae	**3.11**	4.86	**3.77**	**1.74**	**3.95**	**1.87**	3.22
Acari	**3.81**	1.23	0.53	**1.28**	0.64	0.67	1.36
Isopoda	0.45	**5.11**	0.83	0.17	0.15	0.19	1.15
Diptera	0.97	1.9	0.99	0.33	1.13	0.36	0.95
Symphyla	0.74	0	0.02	0.01	0.05	**4.06**	0.81
Hemiptera	0.32	0.96	0.57	0.8	0.12	0.72	0.58
Archaeognatha	0.52	0.46	0.53	0.56	0.54	0.52	0.52
Opiliones	0.93	0.46	0.53	0.2	0.34	0.37	0.47
Diplopoda	0.29	0.27	0.19	0.15	1.06	0.48	0.40
Orthoptera	0.25	0.56	0.36	0.22	0.2	0.08	0.28
Chilopoda	0.05	0.06	0.04	0.25	0.29	0.08	0.13
Oligochaeta	0.11	0.13	0.23	0.09	0	0.14	0.12
Dictyoptera	0.16	0.08	0	0.03	0.02	0.01	0.05
Gastropoda	0.05	0.08	0.08	0	0	0.07	0.05
Pseudoscorpionida	0.13	0	0.08	0.04	0	0	0.04
Lepidoptera	0	0.02	0.04	0.05	0.05	0.02	0.03
Turbellaria	0	0.04	0.11	0.01	0	0.02	0.03
Dermaptera	0.04	0.06	0.06	0	0	0.01	0.03
Nematoda	0.05	0	0	0.01	0	0.01	0.01
Thysanura	0	0	0	0.01	0	0	0.00

MDS (stress = 0.25) analysis did not reveal any differences in the invertebrate assemblages between sites with the high stress value indicating that it is a poor representation of the data (figure 4.1). The ANOSIM results also indicate that the six sites are all similar in terms of invertebrate communities, as the R-value is close to zero (R = 0.066, *p* = 0.1%).

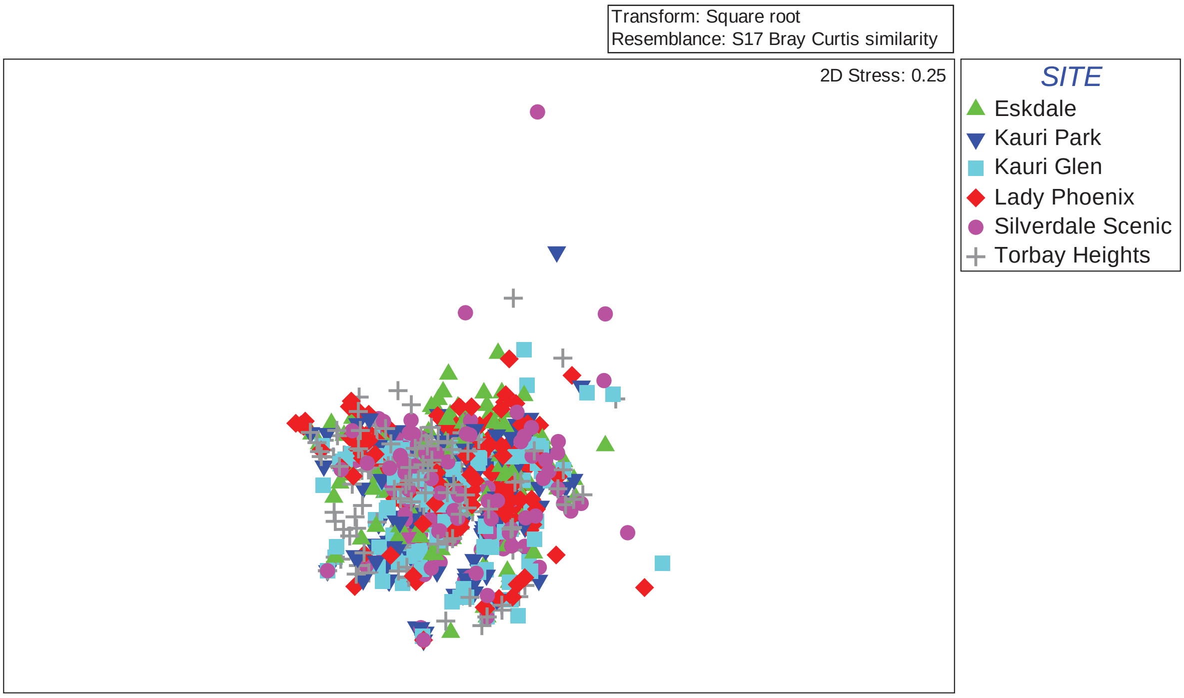

Figure 4.1 - MDS plot showing invertebrate samples from the different sites.

Of the three likely prey groups, Coleoptera was by far the most common at all sites (figure 4.2 and table 4.5). However, the proportion of the total invertebrate captures represented by prey-sized Coleopterans (< 10 mm, see section 3.2) varied from 5.76% at Lady Phoenix Reserve to 17.53% at Silverdale Scenic Reserve. Acari were the least abundant of the likely prey items, except at Eskdale Reserve where they were more abundant than Araneae (table 4.5). Overall, the lowest numbers of likely prey items were found at Kauri Glen and Kauri Park (768 and 706 individuals, respectively), while the largest number of likely prey items was caught at Lady Phoenix Reserve (1164 individuals). Interestingly, although the total number of likely prey items at Lady Phoenix Reserve was high, they made up the lowest proportion of the total invertebrate catch, comprising 8.65% compared to Silverdale scenic Reserve, where the 870 individuals caught represent 21.36% of the total invertebrate catch.

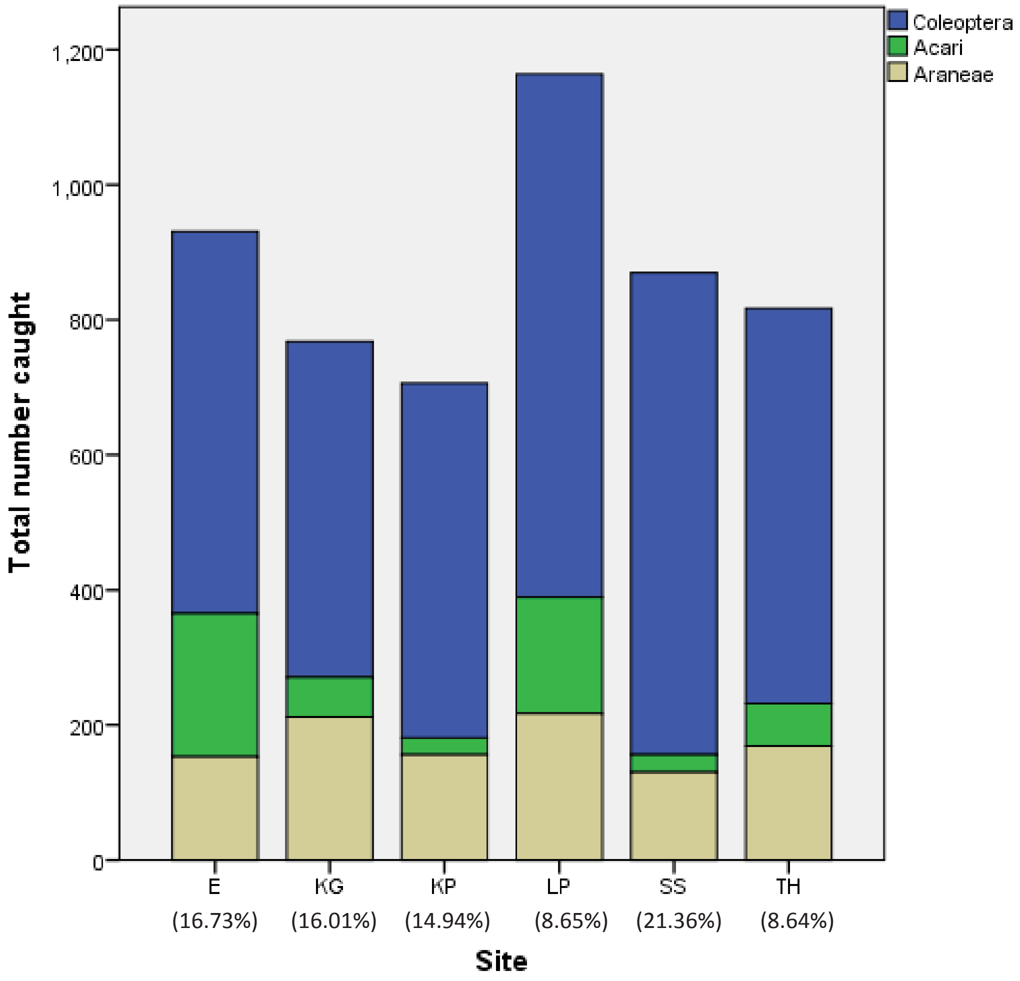

Figure 4.2 - Abundance of likely prey items. Likely prey items are < 10 mm in size and belong to the groups Acari, Araneae or Coleoptera (see section 4.2). Numbers in parentheses are the proportion of total invertebrate captures at each site represented by likely prey items. E = Eskdale Reserve; KG = Kauri Glen; KP = Kauri Park; LP = Lady Phoenix Reserve; SS = Silverdale Scenic Reserve; TH = Torbay Heights.

Table 4.5 - Number of individuals of likely prey groups (< 10 mm in size, size classes 1 and 2) caught at each site.

Site	Coleoptera	Acari	Araneae	TOTAL
Eskdale	566	212	153	931
Kauri Glen	497	59	212	768
Kauri Park	525	25	156	706
Lady Phoenix	775	172	217	1164
Silverdale Scenic	714	26	130	870
Torbay Heights	585	63	169	817
TOTAL	3662	557	1037	5256

4.3.2 – Comparison of invertebrate communities between sampling sessions and baits

Chi-squared tests of homogeneity indicated that there were significant differences in the total number of individuals caught between sampling sessions, but the responses of individual taxa varied (table 4.6 and figure 4.3).

Table 4.6 - Results of chi-squared tests of heterogeneity for the total catch of selected taxa (Acari, Araneae, Coleoptera, Collembola, Hymenoptera) between sampling sessions at each site. Degrees of freedom = 8. Bonferroni correction applied (α = 0.008). Significant results indicated by *.

Site	x^2	*p*
Eskdale	823.08	< 0.001*
Kauri Glen	3925.52	< 0.001*
Kauri Park	446.45	< 0.001*
Lady Phoenix	2759.82	< 0.001*
Silverdale Scenic	1550.79	< 0.001*
Torbay Heights	279.31	< 0.001*

Araneae and Collembola, for example, consistently declined over the three sampling sessions at all sites, while Acari declined strongly at Eskdale and Lady Phoenix Reserves, but increased slightly at the other sites (figure 4.3). The catch of Coleoptera at most sites was fairly stable, with relatively small changes between sessions, except at Silverdale Scenic Reserve where the catch declined dramatically in session two before increasing again. Hymenoptera showed the most variable captures over the three sampling sessions.

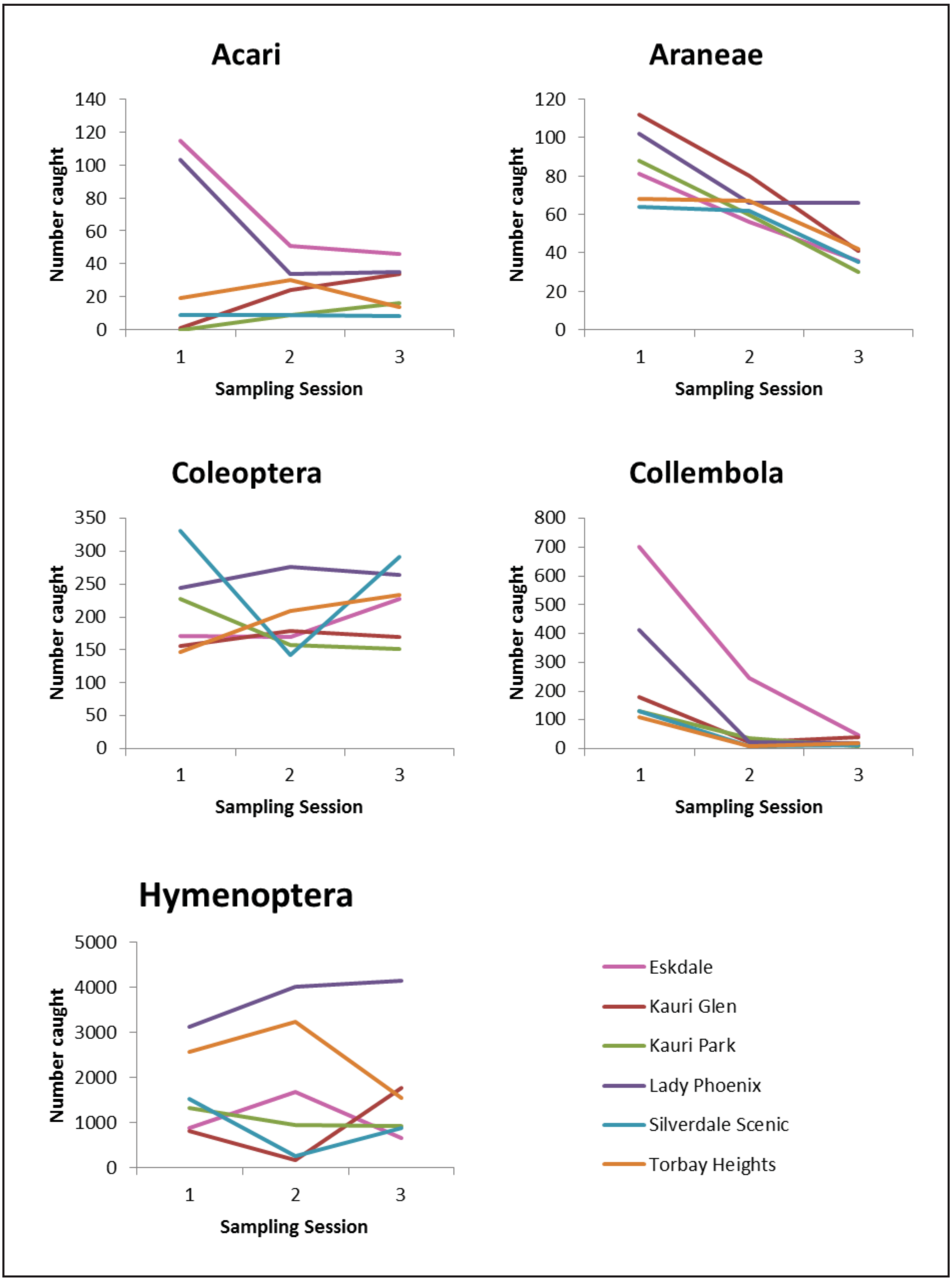

Figure 4.3 – Total catch of selected taxa during each sampling session at each site.

Although the total abundance of individual taxa was variable between sessions, the community composition appeared to be similar, with both MDS (stress = 0.25) (figure 4.4) and ANOSIM (R = 0.069, *p* = 0.1%) indicating that there were no significant differences in the invertebrate communities between sampling sessions.

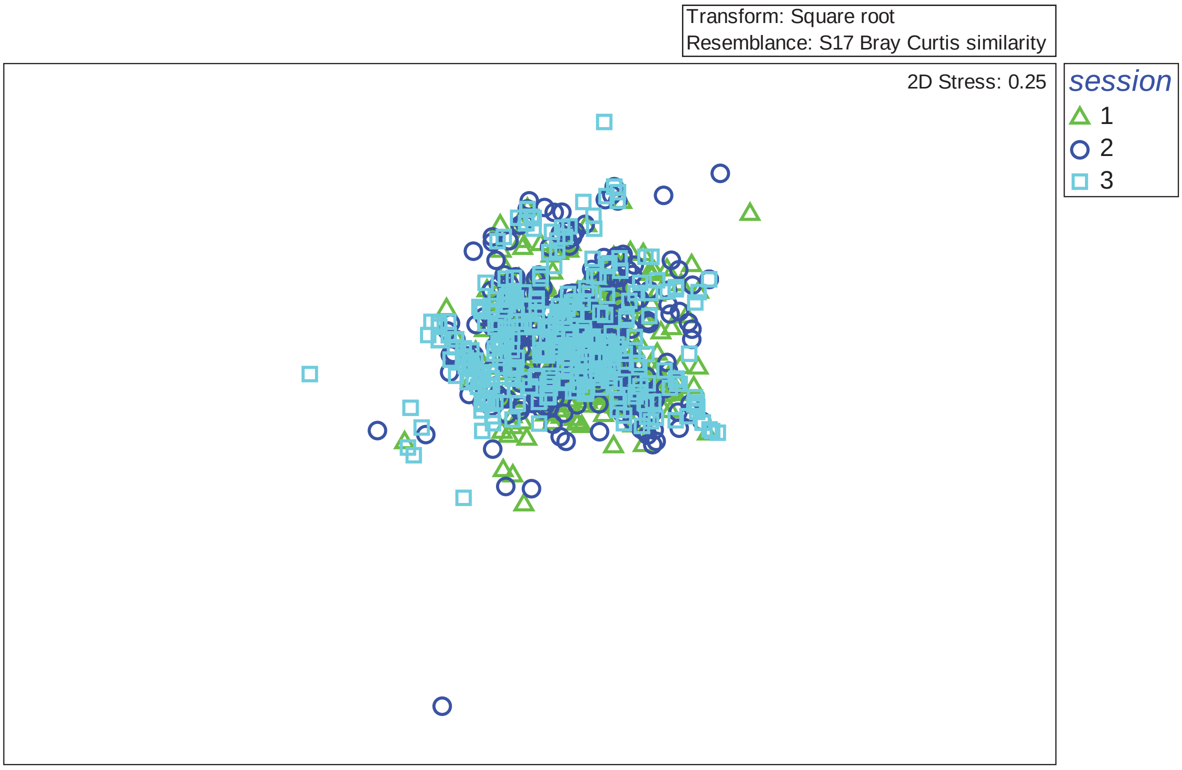

Figure 4.4 - MDS plot showing invertebrate samples from the different sampling sessions.

A chi-squared test of homogeneity indicated that there were significant differences in the total number of individuals caught using different baits ($\chi^2 = 13205.78$, df = 8, $P < 0.001$), but the responses of individual taxa varied. Correlations between the various bait combinations returned significant low to moderate correlations for most taxa, indicating no differences between the baits (table 4.7). MDS (stress = 0.25) (figure 4.5) and ANOSIM (R = 0.14, *p* = 0.1%) also indicated that there were no significant differences between invertebrate assemblages caught using the three baits.

Table 4.7 - Spearman correlations between various bait combinations for selected species. Significant correlations are shaded, α = 0.05.

Species	Preferred bait (%)	Cat food v Banana		Cat food v Control		Banana V Control	
		r	*p*	r	*p*	r	*p*
Acari	Banana (53.7)	0.272	0.004	0.172	0.74	0.407	0.000
Araneae	Cat food (34.3)	0.386	0.000	0.161	0.95	0.395	0.000
Coleoptera	Banana (59.4)	0.521	0.000	0.357	0.000	0.436	0.000
Collembola	Banana (38.5)	0.519	0.000	0.444	0.000	0.572	0.000
Hymenoptera	Cat food (88.9)	0.438	0.000	0.198	0.040	0.264	0.006

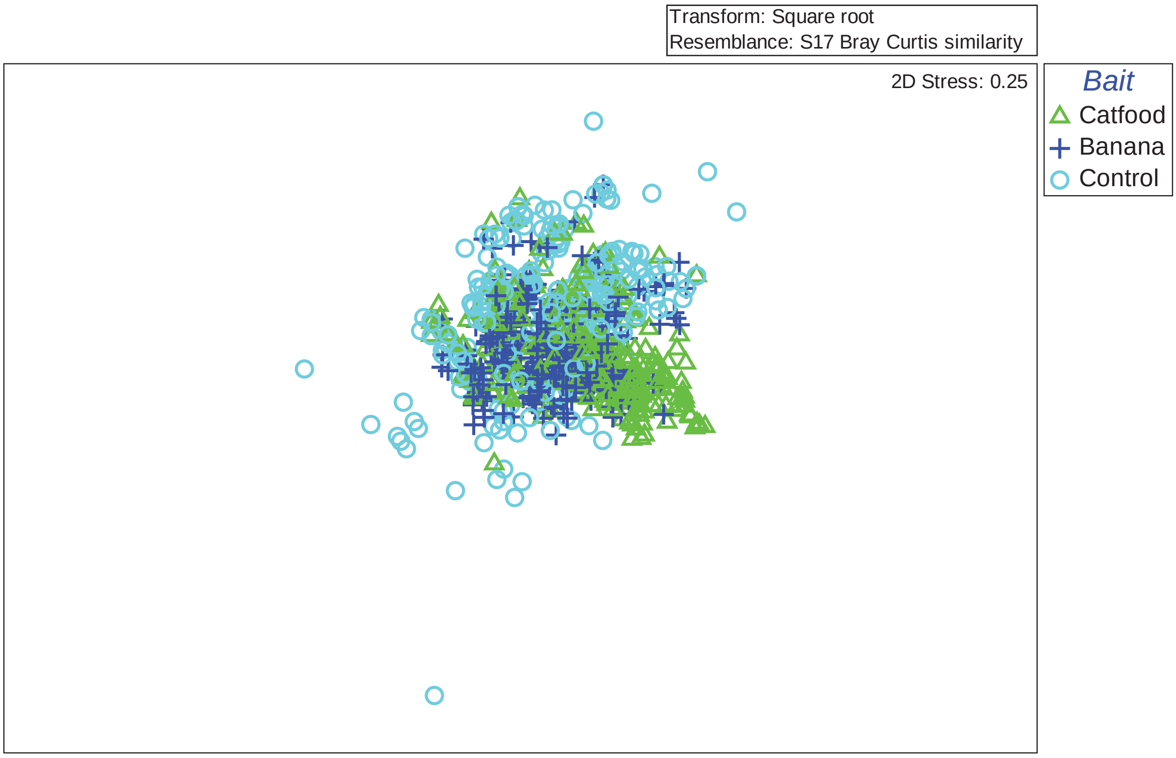

Figure 4.5 - MDS plot showing invertebrate samples from the different baits used in pitfall traps.

4.3.3 – Relationships between invertebrate, native lizard and competitor abundance

Spearman's rank correlation showed that total invertebrate abundance at each site did not correlate with either mammal abundance (r = 0.522, *p* = 0.288) or the abundance of all competitors (mammals and rainbow skinks) (r = 0.371, *p* = 0.468). There was also no correlation between native lizard abundance and the abundance of all invertebrates (r = -.406, *p* = 0.425) or likely food species (r = -.754, *p* = 0.084).

4.4 – Discussion

While the pitfall trap design used in this study was not successful in capturing lizards (see chapter 2), it proved to be highly successful for invertebrates. The diameter and depth of the pitfall traps were in line with the measurements suggested in Woodcock (2005), and each site contained 18 traps for each of the three baits used, which exceeds the recommended minimum number per site of 12 traps. The spacing of 25 m between the traps was sufficient to allow each point to be considered independent of neighbouring points, according to Digweed *et al.* (1995). Another factor contributing to the success of the pitfall trap design was the mesh covering the top, which prevented other animals (probably mammals, although ground-foraging birds such as blackbirds (*Turdus merula*) may also try to access traps) removing baits or invertebrates from the traps (plate 4.2).

Plate 4.2 - One of the pitfall traps which an animal has made an unsuccessful attempt to gain access to.

Because of strict time constraints, invertebrate trapping had to coincide with lizard trapping, which provides several important limitations. Firstly, a solution to kill/preserve trapped invertebrates could not be used as it would also kill any lizards that would be caught. Because of this, there is a potential that larger invertebrates caught in traps may eat some of the smaller ones, leading to the smaller size classes being under-represented. To reduce the incidence of smaller invertebrates being consumed by larger ones, the trapping period was short and traps were checked as soon as practically possible the morning after they had been set. Another key limitation of trapping lizards and invertebrates simultaneously is that any lizards caught in traps would potentially eat some of the invertebrates. To deal with this issue, I intended to use invertebrate data from only the traps where lizards were not caught, but as only two lizards were caught over 1,944 trap nights this was not a significant issue.

4.4.1 – Comparison of invertebrate abundance and diversity between sites

Acari, Araneae and Coleoptera were among the most common taxa caught at all sites in this study (table 4), and these taxa are known to be important in the diets of both copper and ornate skinks, together making up 60% and 48.5% respectively of all food items (including plant matter) (Porter, 1987, summarized in table 4.8). It is interesting to note that I did not catch Gastropoda at Silverdale Scenic or Lady Phoenix Reserves, and no ornate skinks were found at these sites (table 4 and 2.1). Porter (1987) found that shelled mollusca (including Gastropoda) were important in the diet of ornate skinks, making up 15% of all food items, but they were not found in the diets of copper skinks. Eskdale Reserve was the site with the highest captures of both Acari and copper skinks, which is consistent with the results of Porter (1987), who found that Acari was the single most common item in copper skink diets, comprising 35% of all food items. However, because of the extremely low capture rate of lizards in this study, I cannot conclude whether the low number of lizard captures is the result of the methods used being inefficient at detecting the presence of low-density populations, or whether there are no skinks present. Therefore, no definite conclusions about the relationship between invertebrate abundance and lizard abundance can be made. Bearing this in mind, it is interesting to note that Torbay Heights and Lady Phoenix Reserve both had comparatively low proportions of likely prey items (table 4.5) and only low to moderate numbers of native skinks caught (table 2.1). Conversely, Kauri Park had a much higher proportion of likely prey items, and also had the highest skink capture rate of any site.

Table 4.8 - Percentage occurrence of selected food items in the diets of copper and ornate skinks in Auckland grassland reported by Porter (1987).

	Copper skink	Ornate skink
Acari	35	13.5
Araneae	12	10
Coleoptera	13	25
Hymenoptera	8	7.5
Shelled mollusca	0	15

One important factor to consider when comparing my results with Porter’s is that his study was undertaken in grassland habitats, which may have different invertebrate community compositions than would be found in the native forest habitats examined in my study. Porter studied the diets of skinks by examining the contents of faecal pellets, and gave no information regarding the abundance of the various taxa in the environment. However, it is known that generalist feeders (such as copper and ornate skinks) often consume prey items in proportion to their abundance in the environment (Luiselli, 2006; Newbold & MacMahon, 2009; Maritz & Alexander, 2014). Some invertebrates that are common in forest habitats may be rare or absent in grasslands due to the markedly different temperature and humidity regimes, such as litter-dwelling taxa that are highly susceptible to desiccation, such as Collembola and Isopoda. Porter found that these two species made up just 1% and 1.5 % of copper and ornate skink diets, respectively. My study suggests that these species are fairly common in forest habitats (being the third and seventh most common groups overall, table 4), and given that generalist species may eat what is most available to them, these species may make up a larger proportion of the diets of forest-dwelling skinks than those that live in grasslands. As such, any extrapolation of diets between different habitat types must be made with caution.

One limitation of the methods used in this study is that size classes 1 and 2 used to classify invertebrates were too broad to allow detailed comparisons to be made. The partitioning of resources among co-occurring species is a well-known mechanism by which similar species may coexist (Schoener, 1974; Vitt & Decarvalho, 1995). Given the overlapping distributions, habitat use, activity patterns and diets of copper and ornate skinks (Hardy, 1977; Porter, 1987; Gill & Whitaker, 1996), resource partitioning would be necessary for these two species to persist together. While the only published study on the ecology of copper and ornate skinks in the same locality did not find any sizeable sympatric populations, there was some evidence of resource partitioning between the two species in the form of differences in invertebrate prey size. The larger ornate skink took larger items over a greater size range than did the copper skink, and the greatest differences between prey sizes of the two species were seen in hard-bodied invertebrate species which would require a larger, stronger jaw to consume (Porter, 1987). Other studies have also found evidence supporting the idea that similar lizard species may be able to coexist by utilizing prey of different sizes (Schoener & Gorman, 1968; Montoya & Burns, 2007). In a study in the Manawatu region, Gill (1976) found that two sympatric skink species mainly consumed prey items that were 1–5 mm in size, but the two species differed markedly in the proportions of prey items consumed that were < 1 mm, and 5–10 mm in size. As such, size class 2 in the present study (3–10 mm) is possibly too broad and may include items which are too big for the copper skink. This may mask differences in the availability of different-sized prey between sites that could influence skink competition and distributions. I suggest a better range of size classes for future studies would include classes for invertebrates that are <1 mm, 1–5 mm, and 5–10 mm, as per Gill (1976).

4.4.2 – Comparison of invertebrate communities between sampling sessions and baits

Although all invertebrate surveying was conducted within the same season, figure 4.3 and the result of the chi-squared test showed that the abundance of many taxa changed over time, with a decrease in total invertebrate numbers observed. The MDS and ANOSIM results indicate that the community composition did not vary significantly between sampling sessions, suggesting that it is not presence or absence of taxa that changed but rather the relative abundances of certain taxa. One possible explanation for this is the drought that occurred during the summer of 2012–2013 while surveying was taking place. Many invertebrates are susceptible to desiccation, and may reduce their activity or be unable to survive in drought conditions (Paris, 1963; Moore & Francis, 1985; Kaspari & Weiser, 2000).

More hardy species such as Coleoptera are less likely to be affected by drought, which may explain why their numbers did not decrease as much as other taxa, and in fact increased slightly.

A decline in invertebrate abundance or activity would potentially have an impact on lizard food availability, particularly if preferred food items are affected. Of the three likely prey taxa identified in section 4.2, Acari and Araneae both declined over time. Porter (1987) found that Acari and Araneae together make up 47% of the copper skink diet, and as such the food availability for copper skinks may be greatly affected by decreases in invertebrate abundance. This illustrates the importance of multiple sampling sessions, even within seasons, as a single sampling session would have been unable to detect the changes in abundance shown by some important invertebrate taxa. Due to time constraints, this study was limited to just one season (summer), but future studies should endeavour to undertake surveys over all seasons in order to capture seasonal changes in abundance that would affect prey availability for lizards.

Continued sampling over a period of time can have an impact on invertebrate abundance. It is necessary to sample over an extended period in order to reduce random variation in the samples obtained and ensure that rarer taxa are included. However, steps were taken to minimize the effects of extended sampling - invertebrates were counted in the field and released near the point of capture to avoid local depletion of the invertebrate community and the breaks in between sampling sessions were designed to allow the invertebrate community to recover from disturbance (Digweed *et al.*, 1995). As such, it is unlikely that the sampling regime had a major impact on invertebrate abundance over time, and a more plausible cause of the observed reduction in invertebrate abundance is the drought conditions experienced.

The distribution of ants is generally highly clumped, and as a result they may be missed from being caught using non-baited pitfalls, thereby making it impossible to get an accurate picture of the invertebrate assemblage (Greenslade & Greenslade, 1971). The use of baits in the pitfall traps of this study produced the opposite problem, as it attracted ants (potentially from long distances), so that they dominated the invertebrate assemblages caught. However, it would have been inappropriate to exclude ants from analysis because they are known to make up reasonable proportions of lizard diets. This issue was addressed when doing MDS and ANOSIM by applying a square root transformation in order to reduce the influence of the very abundant ants and improve the contribution of less abundant species.

4.3.3 – Relationships between invertebrate, native lizard and competitor abundance

Although no significant relationships were found between the abundance of native lizards and invertebrates (total invertebrates and likely food species), or between the abundance of invertebrates and competitors, the potential effects of food availability on coexistence of native lizards in the presence of mammals should not be disregarded. Once again, the very low capture rates in this study are a serious limiting factor in the reliability of any analyses.

Chapter 5

Summary

Plate 5.1 - Ornate skink (*Oligosoma ornatum*) caught during hand searching at Kauri Park.

The results of my study do not show any clear trends between the relative abundance of native lizards and introduced mammal abundance, however some interesting patterns were revealed. The relatively high rate of tail loss (41%) suggests that these populations are under considerable predation pressure when they coexist with introduced mammals. Some sites followed the patterns of abundance I expected, such as Eskdale Reserve, which had low numbers of introduced mammals and moderate numbers of native lizards, and Lady Phoenix and Silverdale Scenic Reserves, which had high numbers of introduced mammals and relatively low numbers of native lizards. However, there is no conclusive evidence that the abundance of introduced mammals alone determines the distribution of native lizards. This study suggests that both habitat and the abundance of predators are likely to play a role in determining native lizard distributions. Other studies in New Zealand have also found both haitat and predator presence to be important in determining native lizard occurrence, for example, both habitat quality and lower densities of cats were found to be associated with the presence of common skinks (*O. polychroma*) in urban Dunedin (van Heezik & Ludwig, 2012).

The comparison of Kauri Park and Torbay Heights in the current study is particularly interesting because both sites supported relatively high numbers of introduced mammals, and both had high numbers of lizard encounters; however Kauri Park had the highest numbers of identified native lizards, while the majority of skink encounters at Torbay Heights were almost certainly rainbow skinks. This may be due to the habitat at this site allowing native lizards to persist in the presence of relatively high numbers of introduced mammals. Key features of the habitat at Kauri Park compared with Torbay heights include higher canopy cover, lower density of canopy trees, and higher cover of debris (leaf litter and branches/logs) in the lower shrub layer (LSL). The association of native lizards with more highly structured near-ground level vegetation has also been identified in other studies. One study conducted in the South Island of New Zealand from 1996 – 2002 found that lizards were mostly captured in habitats with vegetation cover of 50% or greater (Norbury *et al.*, 2009), while a study in Brisbane, Australia found structural aspects of the habitat such as the presence of woody debris and weed cover were more important for determining the presence of native reptiles species than the floristic composition (Garden *et al.*, 2007).

Food availability in the form of invertebrate abundance does not appear to play a significant role in the coexistence of introduced mammals and native lizards. The total number of invertebrates at each site was not related to the lizard abundance, although in part this may be due to the very large numbers of ants at some sites. The abundance of likely food items, however, may play a role in determining the abundance of native lizards. Furthermore, the abundance of introduced mammals and exotic lizard competitors was not correlated with invertebrate abundance. I found that the composition of invertebrate communities was not significantly different between sites, suggesting that food availability is similar and probably does not play a significant role in the coexistence of native lizards with introduced mammals.

My study suggests that hand searching is the most efficient method for identifying lizards in urban bush fragments both in terms of time and cost. The main disadvantage with using hand searching is the habitat bias, so in cases where this may be a problem ACOs may be a better choice. Hand searching may also be carried out systematically in order to reduce habitat bias, but if transects are not already in place the time required to install them will make it less time-efficient. The pitfall trap design described here is not appropriate for lizards, probably due to the wire mesh used to exclude mammals. Additionally, the traps were difficult and time-consuming to make and install and had to be checked daily. However, in areas where predation risk is not a problem, removing the mesh cover may allow more lizards to be caught and improve the efficiency. Additionally, identification rates from pitfall traps are likely to be higher than from ACOs and hand searching as escapes are less frequent, and pitfall traps do not require as much observer skill to use successfully.

This study shows that tracking tunnels may be an alternative to snap traps for indexing mammal abundance in urban environments, although more in-depth study is required to confirm this. This is in line with another study in New Zealand forest, which found a linear relationship between tracking and trapping rates of ship rats (K. P. Brown *et al.*, 1996). The use of tracking tunnels rather than snap-traps eliminates the risk of trapping or injuring non-target wildlife, pets and the public, which may be important in high-traffic urban bush fragments.

I found little difference in the invertebrate communities trapped using cat food or banana as baits, or the control in pitfall traps. The results of this study suggest that the use of baits is not necessary for sampling most invertebrates. The use of baits caused ants to dominate the captures in the baited traps and may obscure trends in other species. As such, I would not recommend the use of baits for invertebrate pitfall trapping unless it is necessary to include all ant species in samples.

The results of my study suggest that invertebrate abundance probably does not play a significant role in the coexistence of native lizards with introduced mammals, but habitat structure may be important. As such, in future studies I would recommend investing time and effort on surveying habitat structure rather than invertebrate abundance and composition. Another key factor limiting the ability of this study to make conclusions regarding habitat factors which are important for lizard coexistence was the low number of sites surveyed and the low numbers of lizard captured. Surveying more sites will help to clarify patterns in habitats which support larger populations of native lizards in the presence of introduced mammals. Lizard captures may be improved by using appropriate survey methods (section 5.2), and by increasing the duration of surveying. Due to time constraints, my study focussed on sampling in summer only, but future studies should aim to survey in other seasons, particularly spring and autumn when temperatures are often warm enough for lizards to be active.

The results of this study suggest that canopy cover may be important for lizards to coexist with introduced mammals, although it is unclear whether this is due to the effects of the canopy cover itself, or a result of the effect of canopy cover on the input of debris (leaf litter and braches/logs), which was also found to be related to the relative abundance of native lizards. Future management of lizard habitats and re-vegetation efforts should aim to increase the cover and structure of the lower shrub layer, for example, by planting tree species which drop a lot of leaves and contribute to the formation of a dense leaf litter layer.

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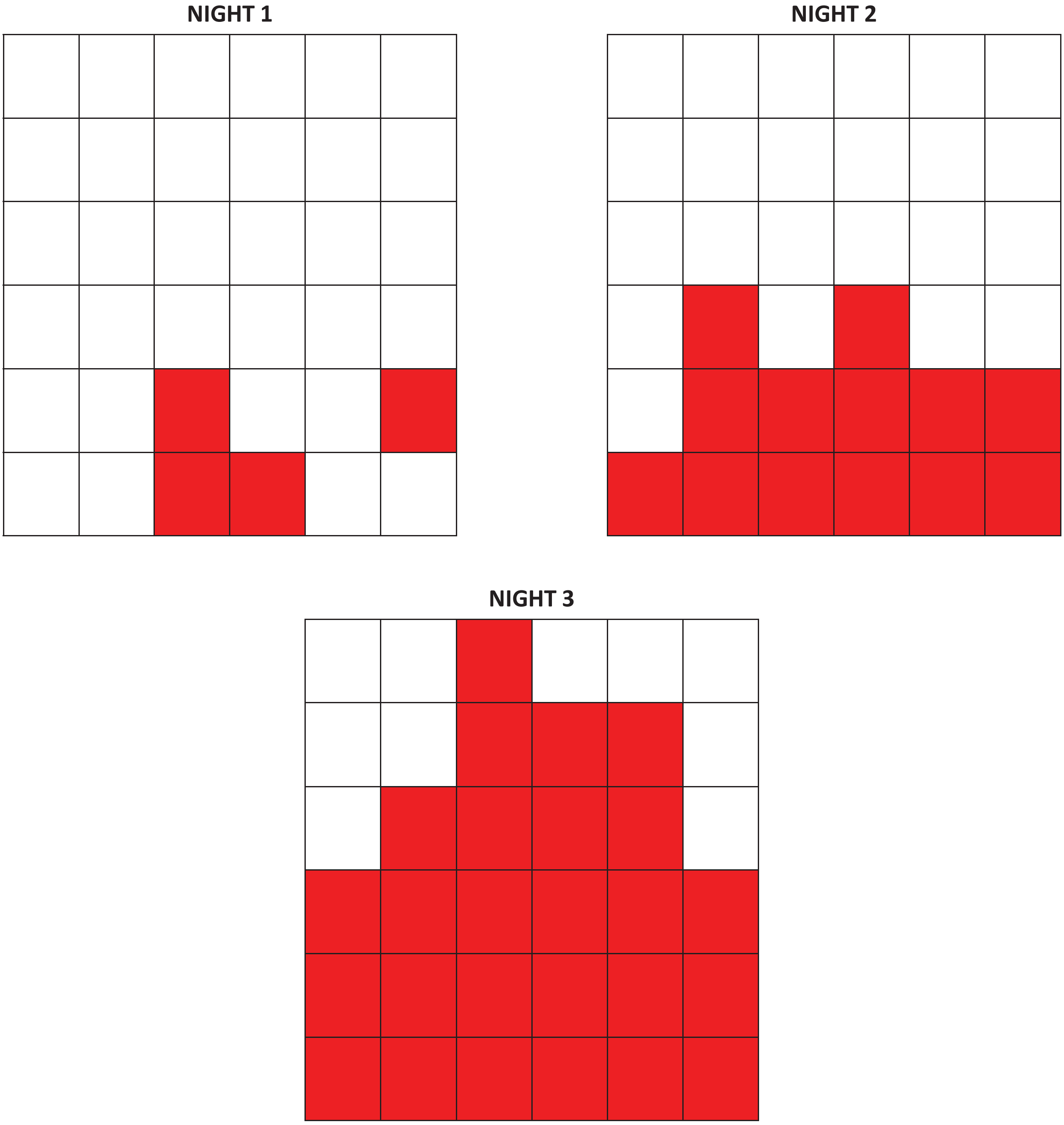
Appendix I

Problems with using baited tracking cards on consecutive nights.

During initial surveys tracking cards baited with peanut butter were used on three consecutive nights at several bush sites within suburban Auckland. It was noted that at most sites, the number of tunnel visited by rats would increase each night in a predictable pattern (see figure 1). It is possible that the rats were able to remember the locations of tunnels found on previous nights, and after visiting these tunnels again and taking the bait they would search for new tunnels. A second potential explanation is that rats were drawn in from outside the area by the scent of the baits but this would not explain the distinctive pattern of tunnel visitation over time. Note that the sampling grid used in this example was 100 metres by 100 metres in size, which was used for initial sampling at some potential sites before being found to be impractical, after which grids were all 50 metres by 50 metres.



NIGHT 1

		🟥			🟥
		🟥	🟥		

NIGHT 2

	🟥		🟥		
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NIGHT 3

		🟥			
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Figure 1 - Pattern of tracking tunnel visitation by rats over 3 nights in January 2012. Each square represents a sampling point, with shaded squares indicating rat tracks.

Appendix II

Table 1 - Numbers of each canopy species at each site as determined by the point-centred quarter method. For a description of the method see section 3.2.1.

Species	Site						All Sites
	Eskdale	Kauri Glen	Kauri Park	Lady Phoenix	Silverdale Scenic	Torbay Heights	
Cyathea	12	14	11	19	31	16	103
Kunzea	14	10	24	9	3	9	69
Phyllocladus	9	16	24	0	9	10	68
Myrsine	12	11	4	4	11	7	49
Melicytus	0	2	0	24	0	0	26
Coprosma	2	0	0	3	6	1	12
Knightia	9	1	1	0	0	1	12
Pseudopanax	4	1	3	0	0	3	11
Leptospermum	1	0	0	3	2	4	10
Geniostoma	0	0	0	5	1	2	8
Pinus	2	0	0	0	6	0	8
Weinmannia	3	0	0	0	0	5	8
Agathis	0	7	0	0	0	0	7
Cordyline	0	1	0	5	0	0	6
Rhopalostylis	0	0	0	0	0	6	6
Dacrycarpus	0	1	0	0	0	4	5
Beilschmiedia	0	2	2	0	0	0	4
Carpodetus	0	2	0	0	0	1	3
Pittosporum	1	0	1	0	0	1	3
Hedycarya	0	1	0	0	0	1	2
Podocarpus	0	0	0	0	2	0	2
Acacia	0	0	1	0	0	0	1
Brachyglottis	0	0	0	0	0	1	1
Cyathodes	1	0	0	0	0	0	1
Dicksonia	0	0	1	0	0	0	1
Dysoxylum	0	1	0	0	0	0	1
Leucopogon	0	0	0	0	1	0	1
Nestegis	1	0	0	0	0	0	1
Olearia	1	0	0	0	0	0	1
Prumnopitys	0	1	0	0	0	0	1
Schefflera	0	1	0	0	0	0	1